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Multimedia Environmental Assessment of Existing Materials Management Approaches for Communities

Office of Research and Development

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US Environmental Protection Agency
Office of Research and Development
Land Remediation and Pollution Control Division
National Risk Management Research Laboratory
Cincinnati, OH, 45268

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Foreword

The US Environmental Protection Agency (US EPA) is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the Agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. To meet this mandate, US EPA's research program is providing data and technical support for solving environmental problems today and building a science knowledge base necessary to manage our ecological resources wisely, understand how pollutants affect our health, and prevent or reduce environmental risks in the future.

The National Risk Management Research Laboratory (NRMRL) is the Agency's center for investigation of technological and management approaches for preventing and reducing risks from pollution that threaten human health and the environment. The focus of the Laboratory's research program is on methods and their cost-effectiveness for prevention and control of pollution to air, land, water, and subsurface resources; protection of water quality in public water systems; remediation of contaminated sites, sediments and ground water; prevention and control of indoor air pollution; and restoration of ecosystems. NRMRL collaborates with both public and private sector partners to foster technologies that reduce the cost of compliance and to anticipate emerging problems. NRMRL's research provides solutions to environmental problems by: developing and promoting technologies that protect and improve the environment; advancing scientific and engineering information to support regulatory and policy decisions; and providing the technical support and information transfer to ensure implementation of environmental regulations and strategies at the national, state, and community levels.

This publication has been produced as part of the Laboratory's strategic long-term research plan. It is published and made available by US EPA's Office of Research and Development to assist the user community and to link researchers with their clients.

Cynthia Sonich-Mullin, Director
National Risk Management Research Laboratory

Executive Summary

The Sustainable and Healthy Communities Program has a mission to develop data and tools that enable community leaders to integrate environmental, societal, and economic factors into their decision-making processes and thus foster community sustainability. This report examines one key area of community sustainability interest, the management of materials from the construction and demolition of buildings, roads, and others structures at their end of life (EOL). Life-cycle assessment (LCA) is an approach frequently used to examine the environmental implications of the EOL management of materials, and while much LCA research has focused on materials from household and commercial community activities (e.g., municipal solid waste), very little effort has focused on construction and demolition debris (CDD). Even though CDD constitutes a substantial volume of material, the role that these materials play with respect to human and ecological health has not been recognized in the same manner as other wastes, and thus they have been less studied.

A meaningful LCA requires a strong database of information (e.g., material makeup and magnitude, energy consumption, environmental emissions) from throughout a material's life cycle. Compilations of such data – a life-cycle inventory (LCI) – provide the backbone for conducting an LCA to examine different materials management strategies. The primary objective of the work presented here was to extensively assess the body of knowledge regarding CDD life-cycle data and to compile US-specific LCI for distinct CDD material categories from publicly available sources. These LCI datasets are intended to complement the existing US EPA LCI database, which includes LCI for a variety of processes and services such as natural resource extraction, manufacturing, energy production, and transportation. An additional objective of this research was to identify data gaps pertaining to CDD LCI and thus identify needed areas of research and information gathering.

LCI were developed for the EOL management perspective of the following CDD materials: asphalt pavement, asphalt shingles, gypsum drywall, CDD wood, land clearing debris (LCD), Portland cement concrete, recovered screened material, and clay bricks. Current EOL management practices were identified based on published industry, government, and academic literature. LCI for each of the CDD materials and several of the associated EOL practices were developed based on input data used by US-specific waste LCA models (e.g., WARM, MSW-DST)

and information included in peer-reviewed and publically available governmental and industry data. For some EOL processes, insufficient US-specific data were available to develop the needed LCI category. For CDD materials recycled in a closed-loop process (e.g., asphalt pavement, gypsum drywall), attempts were made to compile upstream LCI (i.e., those pertaining to product manufacturing), if available.

Although the CDD LCI in this report represent the most comprehensive datasets currently available on this material stream, they are limited because of the relative unavailability of US-specific data; again, CDD has not been examined to the same degree as other EOL materials. Some of the major limitations, and thus identified data gaps, include the following:

i. Liquids emissions associated with the disposal of CDD materials are estimated based on standardized leaching test data, which represent leaching at a liquid-to-solid (L:S) ratio of 20 to 1. These estimates are considered incomplete with respect to the number of constituents and the magnitude of the total emission. Similarly, gaseous emissions are based on methane generation potential data reported for various MSW constituents. These data were used as a proxy to estimate methane and carbon dioxide (biogenic) emission for several CDD materials due to lack of data for CDD materials (e.g., data for branches were used to represent gaseous emission for wood and LCD).

ii. The processing energy requirements for most of the CDD materials are primarily based on equipment manufacturers' specifications compiled by Cochran (2006) due to lack of data from operating processing facilities. The processing LCI developed do not include liquids or particulate matter emissions from handling of CDD materials at processing facilities as these data are not available.

iii. The EOL LCI developed, in general, do not include capital equipment burdens (i.e., emissions associated with the production of infrastructure materials, equipment manufacturing, and energy associated with facility construction) for CDD material management processes; these data were not available in the publicly available literature.

iv. The transport distances between management processes for discarded and processed CDD materials were set at a fixed assumed distance; these data were not available in the publicly available literature.

To complement that data gap analysis, several recommendations for future data gathering and research opportunities are identified and presented in the report.

Acknowledgements and Notice

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Table of Contents

List of Figures

List of Tables

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List of Abbreviations, Acronyms, and Initialisms

ACQ	Alkaline Copper Quarternary
ADC	Alternative Daily Cover
AP-42	Compilation of Air Pollutant Emission Factors
ASMI	Athena Sustainable Materials Institute
BDL	Below-Detection Limit
BOD	Biochemical Oxygen Demand
BTEX	Benzene Toluene Ethylene Xylene
BTS	Bureau of Transportation Statistics
BTU	British Thermal Units
C&D	Construction & Demolition
CDD	Construction & Demolition Debris
CBA	Copper Azle
CCA	Chromated Copper Arsenate
CDRA	Construction and Demolition Recycling Association
CFCs	Chlorofluorocarbons
CKD	Cement Kiln Dust
CLT	Cross Laminated Timber
COD	Chemical Oxygen Demand
CORRIM	Consortium for Research on Renewable Industrial Materials
DCA	Dense Concrete Aggregate
DOT	Disodium Octaborate Tetrahydrate
EAPA	European Asphalt Pavement Association
EIA	Energy Information Administration
EOL	End of Life
ERG	Eastern Research Group, Inc.
FDEP	Florida Department of Environmental Protection
FGD	Flue Gas Desulfurization
FHWA	Federal Highway Administration
FML	Flexible Membrane Liner
ft3	Cubic Foot
GCCS	Gas Collection and Control Systems
GHG	Greenhouse Gas
GREET	Greenhouse Gases, Regulated Emissions and Energy Use
HCFCs	Hydrochlorofluorocarbons
HDPE	High Density Polyethylene
HHD	Heavy Heavy-Duty
HMA	Hot Mix Asphalt
hr	Hour
IE	Impact Estimator
ILCD	International Reference Life Cycle Data
IPCC	Intergovernmental Panel on Climate Change
IVL	Swedish Environmental Institute
IWCS	Innovative Waste Consulting Services, LLC
kg	Kilogram
kJ	Kilojoule
km	Kilometers
kWh	Kilowatt-Hours
kWh/kg	Kilowatt-Hours per Kilogram

L	Liter
lb	Pound
LCA	Life Cycle Assessment
LCD	Land-Clearing Debris
LCI	Life Cycle Inventory
LCS	Leachate Collection System
LCRS	Leachate Collection and Removal System
LEAF	Leaching Environmental Assessment Framework
LEED	Leadership in Energy & Environmental Design
LFG	Landfill Gas
LMOP	Landfill Methane Outreach Program
m	Meter
m2	Square Meter
m3	Cubic Meter
MDF	Medium-Density Fiberboard
µg	Micrograms
mg	Milligrams
MJ/m3	Mega Joules per Cubic Meter
MMT	Million Metric Tons
MOVES	Motor Vehicle Emissions Simulator
MSW	Municipal Solid Waste
MSW-DST	Municipal Solid Waste-Decision Support Tool
MT	Metric Tons
n.d.	No Date
ng	Nanogram
n.r.	No Record
NAICS	North American Industry Classification System
NAPA	National Asphalt Pavement Association
NCSU	North Carolina State University
ND	Not Detected
NMOC	Non-Methane Organic Carbon
NMVOC	Non-Methane Volatile Organic Carbon
NOAA	National Oceanic and Atmospheric Administration
NOx	Nitrogen Oxides
NRC	National Resources Canada
NREL	National Renewable Energy Laboratory
NRMRL	National Risk Management Research Laboratory
O&M	Operation(s) and Maintenance
ORD	Office of Research and Development
OSB	Oriented Strand Board
PAH	Polycyclic Aromatic Hydrocarbons
PAS	Publicly Available Specification
PCC	Portland Cement Concrete
PE	Polyethylene
PM	Particulate Matter
PVC	Polyvinyl Chloride
RAP	Reclaimed Asphalt Pavement
RCA	Recycled Concrete Aggregate
RCRA	Resource Conservation & Recovery Act
RSM	Recovered Screened Material
SHC	Sustainable and Healthy Communities

SOx	Sulfur Oxides
SPLP	Synthetic Precipitation Leaching Procedure
t*km	Tonne-Kilometer
TCLP	Toxicity Characteristic Leaching Procedure
TDS	Total Dissolved Solids
TK	Transfer Coefficient
TNT	Trinitrotoluene
TOC	Total Organic Carbon
Tonne	Metric Ton
TRPH	Total Recoverable Petroleum Hydrocarbon
TSS	Total Suspended Solids
US	United States
US EPA	United States Environmental Protection Agency
USCB	United States Census Bureau
USDA	United States Department of Agriculture
USGS	United States Geological Survey
VOC	Volatile Organic Carbon
WARM	Waste Reduction Model (US EPA Model)
WMA	Warm Mix Asphalt
WRATE	Waste and Resources Assessment Tool for the Environment
WWTP	Wastewater Treatment Plant
XRF	X-ray fluorescence

1 Introduction

1.1 Background

Decision-making processes used by communities to evaluate social, economic, and environmental implications in a resource-constrained world are often not well characterized in terms of the interactions among human health, ecosystem services, economic vitality, and social equity. The need for a decision-making approach that accounts for all three pillars of sustainability (environment, society, and economy) is widely recognized as a key component for transitioning to a more sustainable society (USGS 1998, US EPA 2009, Pereira 2012). The US EPA's Sustainable and Healthy Communities Research Program (SHC) strives to provide tools for community decision-makers to more effectively and equitably weigh and integrate human health and socio-economic, environmental, and ecological factors into their decisions to promote community sustainability. Decisions pertaining to waste and materials management have been identified as one of the highest priorities by communities for implementing sustainable practices (US EPA 2012a). Life-cycle assessment (LCA), in which impacts across the life cycle of a material or process are examined, can be used as a tool to comprehensively assess environmental and human health implications of various available material management options. One of the critical underlying components of LCA are life-cycle inventory (LCI) data, which are a compilation of quantitative inputs and outputs associated with the management of a material (e.g., energy, material properties, and associated emissions and transformations). As part of the SHC research program, the US EPA is developing tools and data that can be used by communities to conduct an LCA for managing various waste and materials management streams during the end-of-life (EOL) phase.

Although computer-based LCA tools have been developed to analyze waste materials and processes, the overwhelming focus to date has been on municipal solid waste (MSW). Another large-volume, non-hazardous materials waste stream, construction and demolition debris (CDD), has largely been excluded from previously-developed models, perhaps owing to the perceived nature of CDD as "inert" or the lack of available data since CDD is often regulated less stringently than MSW in the US. CDD originates from the construction, renovation, repair, and demolition of structures such as residential and commercial buildings, roads, and bridges. Although the composition of CDD materials depends on the nature of the activity (e.g., building construction, building demolition, pavement rehabilitation), wood, asphalt pavement, Portland cement concrete (PCC), masonry, shingles and drywall represent the dominant fractions of CDD materials. CDD materials also contain lesser amounts of such materials as metals, plastics, insulation, cardboard, and soil. In addition, trace quantities of chemical products such as paints, solvents, and adhesives may be present.

Recent estimates suggest that more than 220 MMT of CDD were generated in the US in 2011, and little more than half of this amount of material was recycled. From the disposal perspective, approximately half of the states in the US do not require CDD landfills to be constructed with bottom liners and leachate-collection systems. In light of the large quantity of CDD materials generated annually, the potential for environmental impacts from the disposal of CDD materials and the environmental benefits associated with the recycling and recovery of CDD materials, the US EPA has identified the collection of information and data regarding CDD management practices as a priority area to develop LCI and integrate the collected information into new or improved LCA tools that include CDD management.

An LCA of a material at EOL would account for the energy and material inputs from various unit processes associated with managing the material through final disposition and the associated emissions from that material through management and after final disposition. For example, an LCA pertaining to the recycling gypsum drywall recovered from a building demolition should include the materials (e.g., steel, lubricants) and fuel (e.g., diesel) used by the equipment to grind and screen the drywall, the emissions associated with the production and use of these materials and energy, the emissions occurring during the recycling process itself (e.g., dust emissions), as well as those associated with processing equipment and facility decommission. The LCI data for this example would include the accounting of energy and raw material inputs and emissions to various environmental compartments (i.e., air, water, and land) over the life cycle of the process (i.e., construction, operation, and decommissioning).

1.2 Scope of Work and Objectives

The objective of the research presented in this report was to extensively assess the body of knowledge regarding CDD life-cycle data and compile US-specific publicly-available LCI for the EOL phase for various CDD materials and management processes. Peer-reviewed literature, government and private industry publications, and various LCA modeling tools were reviewed to identify current management practices and the available LCI datasets pertaining to these practices for a set of specific CDD materials. If LCI data were not available, process description and documentation (e.g., included emission categories, background data used to compile the dataset, geographic location. and time period of the data) were reviewed to evaluate the completeness of the dataset. If available, the primary sources of information used to develop the LCI datasets and information were reviewed.

If data on a process were lacking or a given CDD management practice was not in common use, LCI were not compiled. If publicly-available information for a given unit process was not available (e.g., liquids emissions from disposal of CDD), proxy information was reviewed and included as applicable (e.g., CDD materials leaching data). Publicly-available LCI for upstream processes (raw material mining and product manufacturing) were compiled for the materials that are currently recycled in a closed-loop system (i.e., recycled materials are used in the production of the same material). In cases where data gaps existed, LCI developed for non-US conditions were reviewed to better understand the inputs and approaches used to develop such LCI – the unit process LCI that were not available for the US were identified as data gaps in need of further research.

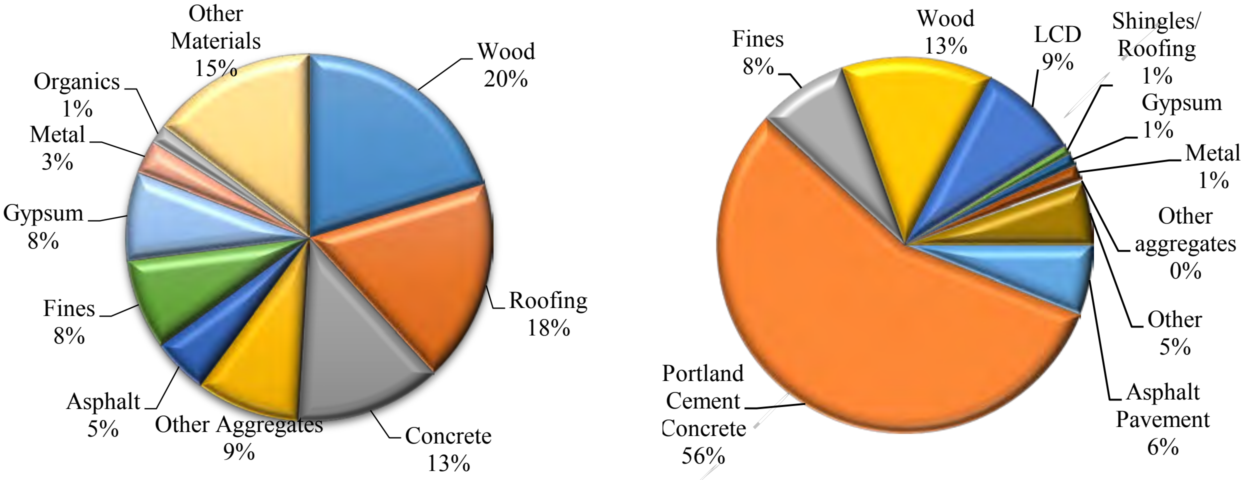
A final objective of this research was to make available the compiled LCI data. The LCI that were developed in this project were formatted to be compatible with the International Reference Life Cycle Data (ILCD) System and were submitted for integration into the existing US EPA LCI database.

1.2.1 Material Types

Based on US EPA (2014) estimates, approximately 88 million metric tons (MMT) of construction and demolition debris (CDD) were landfilled and 47 MMT were recycled by permitted or registered waste management facilities in the US in 2011. Figure 1-1 presents the composition of the CDD landfilled and recycled by solid waste management facilities in the US. The landfilled CDD composition is based on multiple regional waste composition studies, whereas the recycled CDD composition is based on CDD recycled material quantities tracked by Florida, Massachusetts, Nevada, and Washington.

The “other materials” categories mostly consist of indistinguishable and non-CDD materials (about 60%), but also include paper, plastic, glass, carpet, and insulation. “Fines” include materials such as dirt and sand, while “other aggregates” includes bricks, rock, and gravel.

A large fraction of asphalt pavement, Portland cement concrete (PCC), and land clearing debris (LCD) is managed by facilities that do not require solid waste permits, so the composition shown in Figure 1-1 and the previously-mentioned generation estimates do not include large quantities of asphalt pavement, PCC, and LCD. For example, the National Asphalt Pavement Association (NAPA) reported that approximately 72 MMT of reclaimed asphalt pavement were managed by the asphalt pavement industry in 2011 (NAPA 2013). Based on an estimate reported by the Construction and Demolition Recycling Association (CDRA 2014), approximately 127 MMT of PCC is recycled annually in the US.



(a) (b)

Figure 1-1. Composition of CDD (a) Landfilled, and (b) Recycled by Solid Waste Management Facilities in the US

When these additional quantities of asphalt pavement and PCC are considered, PCC, asphalt pavement, wood, roofing materials, fines, gypsum, LCD, and other aggregates constitute more than 95% of the CDD generated in the US. This report presents a compilation of life-cycle inventories (LCI) of various processes needed to conduct an LCA of EOL phase management of the following major CDD constituents:

i. Asphalt Pavement
ii. Asphalt Shingles
iii. Gypsum Drywall
iv. Wood Products
v. Land Clearing Debris (LCD)
vi. Portland Cement Concrete (PCC)
vii. Recovered Screened Material (RSM)
viii. Clay Bricks

The estimate of the total quantity of each material that underwent EOL management in 2011 is presented in each material's respective chapter. These estimates and the sources used to develop the estimates are presented in Table 1-1. The quantities of stockpiled material (i.e., asphalt pavement, asphalt shingles) were excluded from the table. For this report, temporary stockpiling is not considered EOL management.

Table 1-1. Estimated CDD Materials Generation Rate in 2011

Material	Estimated 2011 Quantity Recovered for EOL Management (MMT)	Sources
PCC	212 – 254	CDRA (2014), Turley (2002), Wilburn and Goonan (1998)
Asphalt Pavement	83.5	US EPA (2014), NAPA (2013), Bolen (2013)
Wood Products	24	US EPA (2014)
Asphalt Shingles	10	US EPA (2014), NAPA (2013)
Gypsum Drywall	7.4	US EPA (2014)
RSM	3.7 – 11.9	US EPA (2014), Jang and Townsend (2001a)
Clay Bricks	1.4 – 6.8	US EPA (2014), US EPA (2012b)
LCD	-	N/A

1.3 Report Organization

This report is organized into eleven chapters. Following Chapter 1, Chapter 2 summarizes important details of the materials investigated in this report and then presents LCI information on unit processes that are common to the management of multiple CDD materials targeted in this report, including transportation, primary aggregate production, and environmental emissions (e.g., leachate and gas). Chapters 3 through 10 present material-specific details, including current EOL management practices, an estimate of the quantity of the material managed at the EOL phase (if available), LCI needs and sources reviewed, LCI for the different EOL management processes, and data gaps and opportunities for additional research. The materials examined in these chapters are asphalt pavement, asphalt shingles, gypsum drywall, wood, land-clearing debris, Portland cement concrete, recovered screened materials, and clay bricks. Finally, Chapter 11 summarizes the data gaps identified for the different CDD materials and presents additional research opportunities that would allow the compilation of a complete LCI data search in the future.

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2 Materials and Management Approaches

2.1 Boundary Considered: EOL Phase

The EOL phase, which begins when the material is discarded after the use phase, is the primary focus of the current study. Many LCA studies reported in the literature and from institutional efforts (e.g., LEED Green Building efforts) have focused on building materials (among other CDD) from inception to the point of sale (i.e., cradle to gate), or on the life cycle of their service phase (while in active use). However, the EOL phase is often neglected in LCA studies.

Upstream processes (e.g., raw material extraction, raw material processing) were considered when the material was recycled in a closed loop (e.g., RAP used for asphalt pavement production). However, for materials which are not typically recycled through a closed-loop process, upstream processes were not further explored since these processes would not impact the emissions from the EOL management of the materials. Upstream processes were also considered in some open-loop-recycling cases. For example, dimensional lumber recycled into particle board or PCC recycled as road-base aggregate would include consideration of upstream processes associated with primary wood or aggregate production.

It is important to consider all life phases of facilities and associated equipment used in an EOL management process. For example, while one may expect that the bulk of the environmental burdens from a CDD recycling facility would occur during the operational phase, the emissions associated with construction and decommissioning/demolition of the facility should also be taken into consideration as much as possible. This is particularly important for EOL management-process-dedicated facilities/equipment.

2.2 Geographic Area

The LCI available/used by prominent models/databases were analyzed and discussed in this report but only LCI for the US were compiled in this document.

2.3 Life-Cycle Inventory

An appropriate set of LCI data is needed to conduct an effective LCA, as the results of the LCA are closely tied to the underlying data in the LCI dataset used. LCI datasets should include input and output flows (materials, energy, emissions, etc.) for all processes identified within the materials life cycle (i.e., product system). This section presents terminology frequently used in this report.

A fundamental aspect of a given product or process LCI is the stream of substances, which may be categorized as elementary, product, or waste (see Table 2-1). For a given process, flows are represented and quantified as either inputs or outputs. Flows must be quantified by at least one property, such as volume, area, mass, time, or energy. Flows may be further broadly classified into either the technosphere (flows of material into and from the supply chain or manufactured world) or ecosphere (i.e., belonging to nature) (Schenck 2009). Particulate matter released to the air is an example of an ecosphere flow, whereas the amount of high-density polyethylene (HDPE) needed for landfill construction is a technosphere flow Table 2-1 summarizes and provides examples of the different flow types.

Once a material reaches the EOL phase, the environmental burdens associated with a particular management process are generally quantified by considering the inflows and outflows of energy, materials, and process emissions. Emissions may either be fuel-related (e.g., emissions from fuel extraction, processing, transport, combustion) or non-fuel-related (e.g. dust, leachate); they may originate during facility construction, operation, or decommissioning and may be the result of the production and use of materials that are ancillary to the process (i.e., operation and maintenance consumables) as presented in

Table 2-2. Table 2-2 lists the technical terms used throughout this report to describe the different categories of emissions and materials included in the LCI datasets.

Table 2-1. Basic Flow Categories

Flow Type	Description	Flow Examples for an Example Product System
Elementary	Material or energy entering or exiting the system without prior or subsequent treatment (input or output)	Particulate matter emissions from heavy equipment operation (ecosphere)
Product	Usable or desired material (inputs or output) to another process	Input: Gasoline for combustion in heavy equipment (technosphere) Ouput: Softwood sawn and planed lumber (technosphere)
Waste	Material leaving the product system (output only)	Residuals from CDD processing (technosphere)

Table 2-2. Terminology for EOL CDD Material Management

Term	Description	Example
Pre-combustion Emissions	All emissions released as a result of fuel or electricity production	Air emissions from crude oil extraction, transport and processing for diesel or gasoline fuel production
Manufacturing/ Construction Emissions	Emissions released during the manufacture of a product or piece of equipment/construction of a facility	Dust emissions from land clearing activities for concrete plant construction.
Non-fuel Emissions	Emissions which are not associated with fuel combustion. These emissions are those emitted during a processing step, not as a direct result of energy use or fuel consumption	Emissions from stormwater run-off or landfill leachate to surface or groundwater
Operation and Maintenance (O&M) Consumables	Those materials which are used by a process but are not incorporated into the product of interest.	Lubricating oils for process equipment
Decommissioning/ Demolition Emissions	Emissions released as a result of removing or disposing of process facilities or equipment.	Particulates released as part of material recovery facility demolition
Primary Material	Material produced from virgin resources	Asphalt produced from petroleum refining
Recycled Material	Materials produced from processing of discarded products	Aggregate produced from crushing discarded PCC

Airborne releases of carbon dioxide can either be considered fossil or biogenic. Biogenic carbon dioxide is released due to transformation (e.g., biological decomposition or combustion) of biologically active carbon (e.g., biomass), whereas fossil carbon dioxide is usually released from the combustion of carbon compounds from a fossil origin (e.g., diesel fuel, plastics) or from mineral sources where carbon dioxide would not have been otherwise liberated. Many models and datasets do not consider biogenic carbon dioxide emissions for quantification of emissions associated with human activity as biogenic carbon dioxide emission would occur regardless of human activities (RTI International 2003, US EPA 2012a). However, any anthropogenic-induced increase in greenhouse gas (GHG) emissions from biologically active carbon above and beyond what would have naturally occurred is typically considered in LCA models. For example, the landfill disposal of biomass results in methane emissions which have a substantially greater GHG potency than carbon dioxide. Because it is unlikely that these methane emissions would have naturally occurred without human activity, their impact on the environment would typically be considered in LCA models.

LCI data on emissions to water vary based on assumed environmental controls (e.g., wastewater treatment) and the associated pollutant removal efficiency, which may be quantified on a constituent-specific basis. For example, the MSW-DST assumes removal rates of 97% and 21.6% for aqueous biochemical oxygen demand (BOD) and phosphate, respectively (NC State and Eastern Research Group 2011). Leachate-related emissions for landfill disposal of materials are an example of waterborne emissions considered for material EOL management. LCI data are often available for chemicals normally required to be monitored in leachate on a routine basis by regulations (e.g., BOD, arsenic). Material combustion ash, wastewater treatment plant sludge, and solids collected in air-pollution-control devices are some examples of solid wastes (from CDD materials processing) that should be accounted for in the EOL management of CDD materials. While the amount of solid wastes may be quantified, environmental emissions from these solid wastes and their management are not accounted for in various LCA models (ASMI 2014).

2.4 Organization of Proposed LCI Datasets

All the LCI developed in this report were integrated with the US EPA LCI database using OpenLCA, an open-source LCA program. The flows included in the proposed LCI datasets provided in this report are quantified in terms of a reference flow. All the inputs and outputs included in a LCI are scaled with respect to reference flow. The reference flow is always included under the output flows, but OpenLCA users can switch reference flows for those processes that are producing multiple products of interest. For ease of identification, the reference flow is in italic text in each of the LCI dataset tables presented in the report. All the numbers in the range of 0.0001 to 10,000 are presented in decimal format and the numbers outside this range are presented in engineering notation (e.g., 1.2E-08) for consistency with OpenLCA number format. Also included within the proposed datasets is a column entitled “Category,” which presents the location of flows in the US EPA LCI database as accessible through OpenLCA. All the process LCI developed and presented in this report and the associated elementary flows are included in a folder labeled “Construction and Demolition Debris Management” for ease of identification and review. This folder was created in the “Waste Management and Remediation Services” process folder.

2.5 Common Technosphere Inputs

2.5.1 Transportation

Emissions associated with transportation are often normalized with respect to the amount of material multiplied by the shipment distance, typically expressed as ton-miles (or tonne-km). Ton-miles provide the best single measure of the overall demand for freight transportation services; this measure in turn reflects the overall level of industrial activity in the economy (Dennis 2005). Ton-miles have been historically used by USCB to analyze the magnitude and modes of freight transportation at a national level for different commodities (2007 Commodity Flow Survey by USCB (2010)).

Different LCA approaches and LCI databases quantify transportation-related emissions in different ways. For example, the Municipal Solid Waste Decision Support Tool (MSW-DST), which is a waste-specific LCA tool developed by Research Triangle Institute (RTI) International, North Carolina State University, and the United States Environmental Protection Agency (US EPA), allows for estimates of emissions from transportation for different waste management processes in units of grams of pollutant per ton of waste managed (Curtis and Dumas 2000). Because emission information is already stored in the program for various vehicle types (i.e., grams released per mile traveled), the program requires information regarding three input variables – user-specified material amount, user-specified transport distances, and the number of vehicles necessary for the particular strategy. The total material amount managed is divided by the transport vehicle capacity (which depends on the vehicle type selected) to calculate the number of vehicles needed, which in turn is used to calculate cumulative transport distance. For example, if 1,000 MT of material needed to be transported using trucks with 10 MT capacities, 100 truck trips would be necessary. If each truck needed to travel 20 km per trip, this would result in a cumulative transport distance of 2,000 kilometers.

Similar to MSW-DST, other models estimate the contribution to emissions from both the distance over which materials are transported and the total quantity of materials transported. However, the actual structuring of these calculations is slightly different. Other LCA programs and LCI databases [e.g., WRATE, EASETECH, ECOINVENT, GABI, SIMA PRO, US LCI (2012)] include transportation process units as the product of mass and distance (e.g., tonne-kilometers, ton-miles). For example, both EASETECH and WRATE allow the user to specify the mass/weight of materials transported and the distance that the vehicle travels. Using vehicle-specific default fuel consumption and emission factors, EASTECH estimates the emissions resulting from fuel pre-combustion (i.e., those associated with fuel extraction, mining, processing, transport to the fuel station), and combustion, respectively.

Numerous emissions data in the US LCI database developed by National Renewable Energy Laboratory (US LCI (2012)) LCI database are estimated based on the US EPA's Motor Vehicle Emission Simulator (MOVES). MOVES was developed by the Office of Transportation and Air Quality and uses information from US EPA research; Census Bureau vehicle surveys; Federal Highway Administration travel data; and other federal, state, local, industry, and academic sources (US EPA 2012b). MOVES can simulate time-specific emissions from the operation of a variety of vehicles and vehicle operation stages (e.g. start-up, idling) on a national, state, or county-wide basis. The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model, developed by the Argonne National Laboratory, is also commonly referenced for information concerning the pre-combustion emissions associated with fuel production from initial crude oil extraction to provision at a fueling station.

For quantifying the emissions associated with the general transport of materials discussed in this report, the "Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75" process included in the US EPA LCI database was used to simulate the one-way transport of materials between individual EOL material management locations for distances less than 35 km. The same process, except distinguished as long-haul instead of short-haul, was used to model transport for distances greater than 35 km. For those process datasets where the transport distance is unknown, a distance of 20 km was assumed.

2.5.1.1 Primary Aggregate Transport

Several CDD materials may either contain primary aggregates (asphalt pavement, asphalt shingles, PCC) or may be used as a substitute for the production and transport of primary aggregates as an EOL management option. As a result, it is necessary to develop an estimate of the average nationwide modal (i.e., road, rail, and ocean) distance that primary aggregates typically travel from production to end-use.

The US Census Bureau (USCB) Commodity Flow Survey provides the total amount, distance-amount, and average miles per shipment for gravel and crushed stone in the US. However, the distances and quantities provided are not organized by transport path or end-users (e.g., hot mix asphalt (HMA) plants, PCC plants). Due to lack of end-user-specific data, the average distances for the commodity titled "Gravel and crushed stone" presented in USCB (2010) are assumed to represent the average US-wide aggregate transport distance from production sites to multiple primary (e.g., HMA Plants, PCC plants) end uses for various transport modes. The average distance for a mode was calculated by dividing the total ton-mile data for the mode by the total amount (tons) for all modes combined. Only single-mode transport data provided by the survey were used in the analyses presented in this report.

Table 2-3. Estimated Transport Distances for Aggregate Materials (developed from USCB 2010)

Material Transport	Source and Representative Commodity	Total Amount Transported by Single Mode (million tons)	Mode	Ton-miles (in millions)	Average Transport Distance (miles)	Average Transport Distance (km)
Aggregate Transport from Mine to Paving Mix Plant	USCB (2010)- Gravel and Crushed Stone	1,930	Truck	80,600	41.7	67.1
			Rail	23,400	12.1	19.5
			Water	15,500	8.01	12.9

2.5.2 Electricity

Numerous CDD management processes require electricity for operation, and in many cases electricity consumption is expected to be correlated with the amount of material handled by the process. The current US EPA LCI database contains an “electricity, at industrial user” process, which was used to model the environmental burdens associated with power generation based on the national average grid mix. All the proposed LCI datasets developed as presented in this report use the output flow associated with this process to model electricity consumption since all the datasets represent industrial user electricity demand.

2.5.3 Fuel Combustion in Equipment

Similar to electricity use, multiple CDD management processes require the use of heavy equipment for a variety of tasks (e.g., material loading, unloading, sorting, on-site transport). The current US EPA LCI database process “Diesel, combusted in industrial equipment,” developed by Franklin Associates, was used to simulate pre-combustion emissions and the emissions resulting from diesel combustion for heavy equipment operation.

2.5.4 Other Fuel Combustion Applications

Several processes included in the US EPA LCI database simulate the combustion of other fuels (e.g., gasoline, natural gas, residual fuel oil) in various types of industrial equipment (e.g., boilers). Input flows from these other processes were selected to approximate the emissions associated with combustion of various fuels usage other than equipment operation. For example, the “Natural gas, combusted in industrial boiler” was selected to model natural gas fired at an HMA plant.

2.5.5 O&M Consumables

In addition to direct emissions, LCI datasets should include the emissions associated with the production and use of O&M consumables (e.g. lubricating oils, filters, drilling fluids, belts). While it may not be possible to combine all environmental burdens associated with the production of each O&M consumable, data on the quantity of these materials, if available, were compiled.

2.5.6 Aggregates and Soil

2.5.6.1 Primary Aggregate Production and Fuel Consumption

Several CDD materials incorporate aggregates to increase load-bearing capacity (e.g., PCC, asphalt pavement) and the practice of recycling these recycled materials eliminates the need to produce an equivalent quantity of primary aggregates. It is therefore necessary to understand the fuel-related and non-fuel-related emissions resulting from primary aggregate production.

The United States Geological Survey (USGS) has published annual primary aggregate production and use statistics for crushed stone and construction sand and gravel since 1932 in the *Minerals Yearbook*. A large fraction of these commodities is used in asphalt pavement and concrete production; limestone and dolomite (i.e., calcium carbonate and calcium magnesium carbonate, respectively) are the most common crushed stone aggregates - approximately 70% of all 1.2 billion MT of 2011 crushed stone was limestone and dolomite. Together, bituminous and concrete aggregate account for nearly 65% of all limestone and dolomite sold or used by producers categorized by major end use (Willet 2013). Also, approximately 82% of the 327 MMT of sand and gravel produced in the US in 2011 was used for concrete aggregate and road-related purposes (e.g., road base and coverings, stabilization, asphalt pavement aggregates) (Bolen 2013).

Natural rock is mined and commonly crushed and fractionated at the mining site before shipment (Wilburn and Goonan 1998). Although underground mining of crushed stone is becoming more common due to environmental concerns and better community acceptance, surface methods (e.g. open pit quarries) are the predominant processes for aggregate production (Wilburn and Goonan 1998, USGS 2013). Numerous stages of aggregate processing (e.g., blasting, crushing, screening, size classification, onsite storage) may result in particulate matter emissions (US EPA 2004a). Water consumption information for aggregate mining provided by USGS (2009) suggests that substantial amounts of aggregate mining process water are discharged. Excavated rock is transported to crushers and screening equipment for size-reduction and classification. Pre-combustion (i.e. those emissions released from fuels used for extracting, refining/processing, and transporting) and exhaust emissions are associated with the use of energy for processing equipment operation.

The 2012 US LCI database has a “Limestone, at mine” process developed by Franklin Associates from information compiled from a variety of sources dated from 1998-2004. However, based on the process inputs/outputs, it appears that only fuel consumption and particulate emissions associated with mining and crushing were taken into account. As modeled, the process appears to be missing emissions from the manufacturing and detonation of the explosives used for blasting and does not include any emissions to surface water. The US EPA’s Waste Reduction Model (WARM) uses information from this process from the 2009 version of the US LCI (2012) database to develop the estimate for emissions related to aggregate mining (US EPA 2012a). The GaBi LCI database appears to have a US-specific limestone production process; however, it exclusively references German- and Swiss-published sources in its process documentation page.

The US EPA’s Compilation of Air Pollutant Emission Factors (AP-42) provides air emissions information for crushed stone and sand and gravel processing and for quarrying/mining-specific explosives detonation (US EPA 2004a, US EPA 1995b, US EPA 1980). Although AP-42 (US EPA 1983) includes emissions associated with mining and quarrying blasting agent denotation, the emissions associated with manufacturing blasting agents commonly used in quarrying/mining operations are not included. Persson et al. (1993) reported that approximately 0.4 kg of explosives is needed for loosening a cubic meter of rock. Mehrkesh and Karunanithi (2013) estimate the power consumption for the production of 2, 4, 6,-trinitrotoluene (TNT) as 2.6 kWh/kg. While TNT is not typically used as a blasting agent in quarrying operations, the magnitude of the energy requirement for TNT production demonstrates the importance of including emissions associated with manufacturing quarry-use-specific blasting agents in aggregate production LCI.

Sand and gravel mining does not typically involve blasting. Unlike crushed stone production, sand and gravel mining and processing has numerous sources of process water emissions (e.g., wet screening, log washers, rotary scrubbers, water classification, wet dust suppression, dewatering) (US EPA 2004a, US EPA 1995b). AP-42 presents only air (particulate matter emissions) from sand and gravel processing (US EPA 1995b). The water requirement and the associated liquid emissions to surface and groundwater are not included in any of the LCI sources reviewed. USGS (2009) estimates that 125 to 4,160 liters of water is used for every MT of non-metallic crude ore produced from mining or quarrying; this suggests that aggregate production has a significant water demand and releases substantial amounts of process water for either treatment or discharge to surface water bodies.

Both Stripple (2001) and Ecoinvent contain LCI datasets simulating non-US aggregate production processes, although Stripple (2001) only includes emissions associated with energy consumption and maintenance vehicle operation. Similar to US LCI (2012), Ecoinvent has a limestone production process to model aggregate production, but also includes processes for gravel and sand quarry operation, limestone quarry construction, and gravel and sand quarry construction.

As none of the existing aggregate production LCIs evaluated includes all the emissions discussed above, the “Limestone, at mine” process included in the US LCI (2012) database to model aggregate production was selected for use in the LCI presented in this report, as needed. Future efforts should consider quantification of emissions associated with blasting agents manufacturing and use in mining operations, mining/processing equipment manufacturing and maintenance, surface water emissions, and water consumption. As presented below, the energy requirements estimated from USCB (2001) for producing different aggregates are similar in magnitude to the “Limestone, at mine” process.

Fuel-use LCI data for aggregate mining and quarrying were developed from the 1997 Economic Census Mining Subject Series (USCB 2001) based on total fuel-consumption data for the production of crushed and broken limestone, crushed and broken granite, other crushed and broken stone, and construction sand and gravel. Industry data for the production of each of these aggregates are organized by the North American Industry Classification System (NAICS) code within USCB (2001). The 1997 codes describe the aggregate production industries for limestone, granite, and other crushed and broken stone to include establishments primarily engaged in developing the mine site, mining and quarrying the specific aggregate and related rocks, and preparing the raw ore for use (e.g., pulverizing, grinding) (USCB 2013). The 1997 NAICS code for construction sand and gravel describes the mining industry as including one or more of the following: pit operations; dredging operations; and washing, screening, and other preparation operations for material production (USCB 2013).

Limited data were available in the economic surveys of 2002 and 2007; the data reported in 1997 were used for estimating fuel consumption per unit aggregate production. The total aggregate production for each of the four aggregate categories reported by USCB (2001) for 1997 was used to estimate energy consumption per kg of aggregate production (with the exception of construction sand and gravel). Construction sand and gravel production data were instead taken from the 1997 USGS *Minerals Yearbook* because approximately 40% of the total value of shipments of construction sand and gravel as provided by USCB (2001) was not provided for all categories of materials. In addition to electricity use and consumption of the five specific fuels listed by USCB (2001), there are two other categories of unclassified fuel use: “other” (e.g., wood, coke, liquefied petroleum gas) and “undistributed.” Together these two fuel categories represent from 18% (i.e., crushed and broken granite) to 59% (i.e., construction sand and gravel) of a specific aggregate’s total fuel delivery cost. However, the actual fuel quantity was not published for these two categories of fuels; only the fuel delivered cost was provided. Distillate fuel makes up the majority (i.e., 64-75%) of the total categorized fuel delivery cost for each of the four aggregate categories. Therefore, "other" and "undistributed" fuel consumption was quantified as distillate fuel oil. This was done by multiplying the delivery cost of these fuels with the ratio of the distillate fuel oil consumption to the distillate fuel oil delivery cost.

The approach for estimating material production fuel consumption LCI data outlined above was used by the US EPA (2003) to estimate emission factors for primary aggregate production. Table 2-4 presents the estimated fuel consumption per kg mined and quarried aggregate (by specific aggregate type). For comparison purposes, Table 2-4 also lists the fuel consumption data used for the "Limestone, at mine" process included in the US LCI (2012) LCI database.

Table 2-4. Fuel Consumption per Kilogram Mined and Quarried Aggregate (USCB 2001 and Bolen 1997)

	Fuel Consumption (fuel unit/kg aggregate production)				
Fuel Category	Granite (Crushed and Broken)	Construction Sand and Gravel	Other Stone (Crushed and Broken)	Limestone (Crushed and Broken)	US LCI (2012) "Limestone, at mine" Process
Coal (kg)	W[1]	W	-[2]	4.49E-5	3.58E-5
Distillate Fuel Oil (L)	0.000614	0.000700	0.000835	0.000752	0.000584
Residual Fuel Oil (L)	0.000117	5.26E-5	4.63E-5	5.65E-5	NP[3]
Natural Gas (m^3)	W	4.16E-5	0.000246	7.50E-5	0.000140
Gasoline (L)	3.10E-5	2.27E-5	2.75E-5	4.93E-5	5.11E-5
Electricity (kWh)	0.00323	0.00266	0.00350	0.00365	0.00423

[1] "W" denotes fuel consumption quantities were withheld to protect individual company data
[2] "-" denotes a value of 0
[3] "NP" denotes that the value was not provided

The US LCI (2012) process was developed based on data provided in Franklin Associates. It can be seen that USCB-based estimates for limestone distillate fuel and coal consumption were greater than those used in the US LCI (2012) dataset. However, natural gas, gasoline, and electricity estimates were lower than the respective US LCI (2012) estimates. For comparison purposes, diesel consumption as provided in the US LCI (2012) process was listed as distillate fuel oil. The source document cited by Franklin Associates could not be located to explore the causes of the differences between the estimates presented above and the US LCI (2012)'s data. While coal consumption data were withheld for crushed granite and construction sand and gravel production, the fact that this fuel category was withheld (which suggests that the use of this fuel was limited to a small number of producers), coupled with the fact that this fuel category was not even included in the 2002 Economic Census for these industries, suggests that coal consumption is insignificant in the production of these aggregates. Natural gas consumption data for crushed granite production were not found for other Economic Census years.

Energy content values from the US Energy Information Administration were used to translate fuel consumption quantities (mass and volume) into a common energy unit (kJ) to compare the cumulative energy demand for producing the different aggregate types. The total fuel energy consumption per kg mined and quarried aggregate (by specific aggregate type) is presented in Figure 2-1.

The energy associated with blasting agent and water use is not quantified in the data presented in Figure 2-1. The total fuel energy consumption for producing the different aggregates is similar across the different aggregate categories. As can be seen, the total fuel energy consumption per kg limestone production developed from USCB (2001) is comparable with the US LCI (2012) "Limestone, at mine" process.

2.5.6.2 Natural Soil Production and Fuel Consumption

Similar to the use of recovered CDD materials as recycled aggregate, several of the materials analyzed (e.g. RSM, concrete, asphalt pavement, clay bricks) may be recovered and beneficially used as general fill (i.e., a soil substitute). The LCIs for primary soil production are needed for a comparative LCA of EOL management. The energy-related LCIs were developed for natural soil production using production rate and fuel consumption data for the equipment commonly used for soil production.

CAT (2006) reported typical cycle times and bucket sizes for 12 excavator models. These cycle times and bucket sizes were used to estimate volumetric soil production rates. The volumetric production rate was converted to a mass-based estimate assuming a soil density of 1,330 kg/m^3 soil (USDA 2013). These production rates were used in conjunction with the median value of the medium diesel consumption rate range for each excavator model to estimate the diesel consumption per ton of soil excavated. The average diesel consumption rate among the models is approximately 6.54 × 10^{-5} L/kg excavated soil and was estimated according to the following equation:

$$F_{es} = \frac{1}{n\rho_s}\sum_{i=1}^{n}\left(\frac{F_i C_i}{60B_i}\right)$$

Where,

F_{es} = average diesel consumption rate per unit weight of excavated soil (L/kg)
$Ρ_s$ = soil density (kg/m^3)
B_i = the bucket volume of the ith excavator model (m^3)
C_i = the cycle time of the ith excavator model (minutes)
F_i = median diesel consumption rate for the ith excavator model (L/hour)
n = total number of excavator models

The amount of energy required to excavate natural soil is substantially less than the amount of energy necessary to mine and process (e.g., crush, fractionate) different aggregates. As shown in Figure 2-1, soil production energy is about 2.53 kJ per kilogram of soil, or approximately 5% of the energy needed for aggregate production.

Ecobalance (1999) estimates that more than 80% of municipal solid waste landfill (MSW) sites that use cover soil acquire it from on-site sources. Thus, it is necessary to develop an LCI process dataset that simulates the on-site transport of primary cover soil for CDD materials disposed of at MSW landfills. Data on the heaped capacity and diesel consumption rate for six articulated truck models included in CAT (2006) were used in combination with the average excavator soil production rate approximated above to estimate the diesel consumption resulting from on-site cover soil transport. The load time for each truck model was summed with the round-trip travel time, assuming an average truck speed of 15 km/hr over a 2-km round-trip distance. The average fuel consumption of all the articulated truck models was 1.88 × 10^{-7} L diesel per kg of excavated soil and was estimated according to the following equation:

$$F_{ts} = \frac{1}{m\rho_s}\sum_{j=1}^{m}\frac{F_j}{H_j}\left(\frac{D}{V} + \frac{H_j}{\frac{1}{n}\sum_{i=1}^{n}\frac{60B_i}{C_i}}\right)$$

Where,

F_{ts} = average diesel consumption rate per unit weight of onsite transported soil (L/kg)
m = total number of articulated truck models
D = round-trip distance for onsite cover soil transport (km)
V= average truck speed (km/hr)
B_i = the bucket volume of the i^{th} excavator (m^3)
C_i = the cycle time of the i^{th} excavator (minutes)
n = total number of excavator models
F_j = median diesel consumption rate for the jth articulated truck model (L/hour)
H_j = heaped capacity of the jth articulated truck model (m^3)
ρ_s = soil density (kg/m^3)

Information on particulate emissions resulting from soil excavation and on-site transport was not found. Ecobalance (1999) found that the average cover soil travel distance from off-site production to MSW landfills within the US and Europe was approximately 8 km. This is the distance assumed for off-site cover soil transport for CDD landfills in this report's analysis.

2.5.6.3 Primary Aggregate Production Particulate Emissions

The US EPA (1995a) published particulate matter emission factors for crushed stone and pulverized mineral processing and sand and gravel processing. Particulate matter emission factors are provided for various steps of aggregate processing; however, separate emissions factors are published for controlled/uncontrolled processing steps (e.g., crushed aggregate screening (controlled) versus crushed aggregate screening (uncontrolled)). Sand and gravel processing emission factors are only provided for industrial sand and gravel (which is processed differently than construction sand and gravel); because of an absence of additional data, US EPA (1995b) recommends applying the emission factors published in the crushed stone and pulverized mineral processing section for modeling emissions from construction sand and gravel processing. The sum of uncontrolled emissions from all steps in processing crushed stone was used to provide a total conservative particulates air emission factor for both crushed stone and sand and gravel production processes for the following reasons:

- No data were available for emissions from primary and secondary crushing – emission factors for these steps are not published.
- Background documentation provided in US EPA (2004) does not discuss the distribution of controlled versus uncontrolled steps in the crushed stone processing industry.
- Emission factor ratings for the processing steps as provided in US EPA (2004) are typically identified as below average and poor.

US EPA (2004) also categorizes uncontrolled aggregate processing particulate emissions into particulates smaller than 10 microns. This specific particulate category was separately used for developing particulate emission factors (i.e., PM<10, PM>10). AP-42 does not quantify the particulate emissions resulting from aggregate blasting, excavation, transport on haul roads, or emissions from aggregate stockpiles, and (in addition to primary and secondary crushing as noted above) no data were available for particulate emissions resulting from wet drilling and truck loading/unloading of aggregates, as shown in Table 2-5.

Table 2-5. Uncontrolled Particulate Matter Emissions from Crushed Stone and Pulverized Mineral Processing (US EPA 2004)

Source	PM > 10 microns (g/kg Crushed Stone)	PM < 10 microns (g/kg Crushed Stone)
Primary Crushing	ND	ND
Secondary Crushing	ND	ND
Tertiary Crushing	0.0015	0.0012
Fines Crushing	0.012	0.0075
Screening	0.0082	0.0043
Fines Screening	0.114	0.036
Conveyor Transfer Point	0.00095	0.00055
Wet Drilling - Unfragmented Stone	ND	4E-5
Truck Unloading - Fragmented Stone	ND	8E-6
Truck Loading - Conveyor, crushed	ND	5E-5
Total	**0.0870**	**0.0496**

[4] "ND" denotes that no data were available.

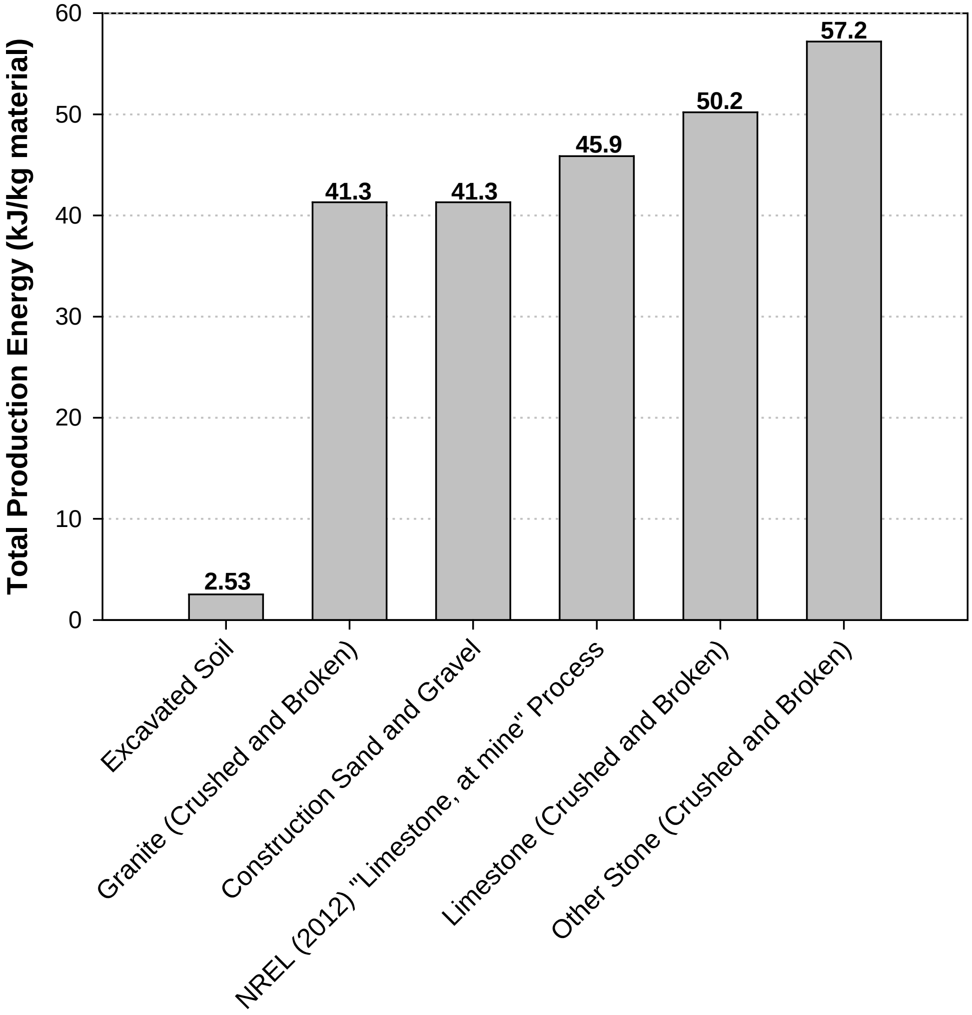

Figure 2-1. Total Fuel Energy Consumption per Kilogram Production of Aggregate and Soil (developed from USCB 2001, Bolen 1997, and CAT 2006)

2.5.6.4 Recycled Aggregate Production Fuel Consumption

Energy consumption for processing demolished PCC and reclaimed asphalt pavement (RAP) is provided by Wilburn and Goonan (1998), based on data from an energy audit of a material recovery facility in Denver (Colorado). However, the energy requirements for these materials are not categorized by fuel type (e.g., electricity, diesel). Therefore, an energy feedstock mix as 50% diesel and 50% electricity (same as that used by US EPA (2003) to develop the dataset for the production of recycled concrete aggregate) was assumed. US EPA (2004) does not provide an estimate of particulate emissions resulting from the primary or secondary processing of crushed stone; due to this lack of data, particulate emissions associated with processing of these recycled materials could not be estimated. Energy consumption data for processing both demolished PCC and RAP are provided in Table 2-6.

Table 2-6. Energy Consumption for Processing Demolished PCC and RAP

Energy	Unit	Energy Consumption (per kg Material)	
		Demolished PCC	RAP
Diesel	L	0.000440	0.000213
Electricity	kWh	0.00472	0.00229

2.5.6.5 Data Gap Analysis of Primary Aggregate Production

Table 2-7 summarizes LCI data identified for primary aggregate production. Only the GaBi and US LCI (2012) databases provide LCI datasets for at least one type of aggregate production. WARM and AP-42 only provide greenhouse gas and particulate matter emissions, respectively, while Wilburn and Goonan (1998) only discuss the energy required for processing primary aggregates.

Table 2-7. Overview of Aggregate Production LCI Data Available

Process	WARM	GaBi	AP-42	Wilburn and Goonan (1998)	US LCI (2012)
Aggregate Production	P	X	P	P	X

"P" and "X" denotes partial, and most comprehensive dataset currently available, respectively. These notations are used throughout the report.

The following data gaps were identified after reviewing the extent of available US-based LCI information on primary aggregate production:

1. **Emissions information associated with blasting agent manufacture and detonation at quarries.** AP-42 (US EPA 1980) data on emissions resulting from explosives detonation could be combined with rock constant information (Persson et al. 1993) and a representative in-place density of unmined aggregates to approximate blasting emissions on a per-mass-mined basis. However, additional information from explosives manufacturers would be necessary to estimate the emissions associated with the manufacture of quarry-specific explosives.

2. **Emissions from mining/processing/grinding equipment manufacturing.** For a complete LCI of the environmental burdens associated with aggregate production, it is necessary to quantify the material, energy, and emission burdens associated with manufacturing all cradle-to-gate aggregate production equipment. These data do not appear to be available.

3. **Water emissions/consumption information from quarry work.** Although USGS (2009) provides an estimate of the range of water necessary for producing raw aggregate ore, specific process water emissions associated with the production of various aggregate types were not located. Based on the range provided by USGS (2009) (125-4,160 L water per MT crude ore), it appears the aggregate production process may result in substantial discharges of suspended and (potentially) dissolved solids to surface water.

2.5.7 Mixed CDD Processing

Calhoun (2012) discusses the mass fraction of recovered materials and diesel consumption resulting from the operation of five Florida CDD processing (i.e., recycling) facilities for 2011. Diesel consumption on a per-kilogram-material-processed basis was estimated as 0.00199 liters by dividing the total amount of diesel used by all the CDD processing facilities by the total mass of material received by all the CDD processing facilities according to the following equation:

$$D_F = \frac{\sum_{i=1}^{n} D_i}{\sum_{i=1}^{n} M_i}$$

Where,

D_F = diesel consumption per kilogram of CDD processed (L/kg)
Di = total diesel consumption at the ith CDD processing facility for 2011 (L)
M_i = total mass of material received at the ith CDD processing facility for 2011 (kg)

Similar equations were used to estimate fossil fuel consumption for the management of different materials throughout the report. The electricity consumption on a per-material-processed basis was estimated similar to the diesel consumption, but the estimate only used electricity billing data from a single north Florida CDD processing facility. The south Florida CDD processing facilities did not provide the purchased electricity cost even though this information was requested by Calhoun (2012). The total amount of electricity used by the north Florida facility was estimated using the total 2010 electricity purchase amount (i.e. $109,272) and the average 2010 retail price of electricity sold to the Florida industrial sector from EIA (2014) of $0.0885/kWh. Approximately 1.23 million kWh of electricity consumption is estimated for 2010.

Calhoun (2012) organizes processed material outputs into 7 categories: wood, RSM, yard waste, concrete, metal, cardboard and miscellaneous (waste residuals are assumed as the difference between the total mass of received material and the sum of the processed material outputs). However, the north Florida facility provided 4 additional categories of recovered material instead of the "miscellaneous" category reported by the south Florida facilities: shingles, plastic, glass and textiles. For the purpose of developing a CDD processing facility LCI dataset, it was assumed that the "miscellaneous" material recovered by the south Florida facilities included these four materials at the same percentages as the north Florida facility.

The estimated diesel and electricity consumption for CDD processing facilities is approximately 1.99 milliliters and 0.025 kWh per kilogram of processed material. Figure 2-2 shows the estimated mass-fractional representation of materials recovered or discharged from a CDD processing facility. Compressed gas is another fuel described in Calhoun (2012) which is used at the CDD processing facilities for forklifts and floor-sweeping units, but a fuel consumption estimate was not provided. Because CDD processing facilities are not operated to recover residuals or RSM, these two material output flows were assigned as waste flows in the LCI dataset. Therefore, diesel and electricity consumption are not allocated to either residuals or RSM, and these materials are produced burden free to downstream processes.

Based on these material fraction output and diesel consumption estimates, Table 2-8 presents the proposed LCI dataset for a CDD processing facility. Although no electricity requirement estimate was found, this input flow was included as a placeholder. No nationwide average transport distance was found for mixed CDD movement between the point of generation and a mixed CDD processing facility; a distance of 20 km was assumed.

In addition to US-data, additional LCI information was found in Ecoinvent documentation. Doka (2009) discusses the fuel and electricity consumptions associated with sorting and crushing building materials, which includes various CDD components (e.g., wood, glass, bricks, concrete). As described in Doka (2009), building material sorting plants typically use an up-front screening/separation process to remove fines and bulky items such as metals, wooden poles, and windows. This separation step is followed by a crushing/sorting step for larger materials such as concrete and brick. Based on this review, Doka (2009) assumes a total electrical usage of 3.7 kWh/MT of building material where 1.5 kWh is the electricity usage estimated solely for crushers alone while the remaining 2.2 kWh/MT was assumed to be the electricity usage for other sorting machinery.

Diesel fuel consumption occurs at CDD processing facilities from heavy equipment used to unload, sort, transport, and load waste at facility tipping floors. The Doka (2009) diesel fuel consumption estimate for fuel usage was also based on literature-derived values corresponding to the fuel demand of a skid-steer loader; values ranged from 2.95 - 5.9 MJ/m^3 of sorted building waste with an average value of 4.4 MJ/m^3.

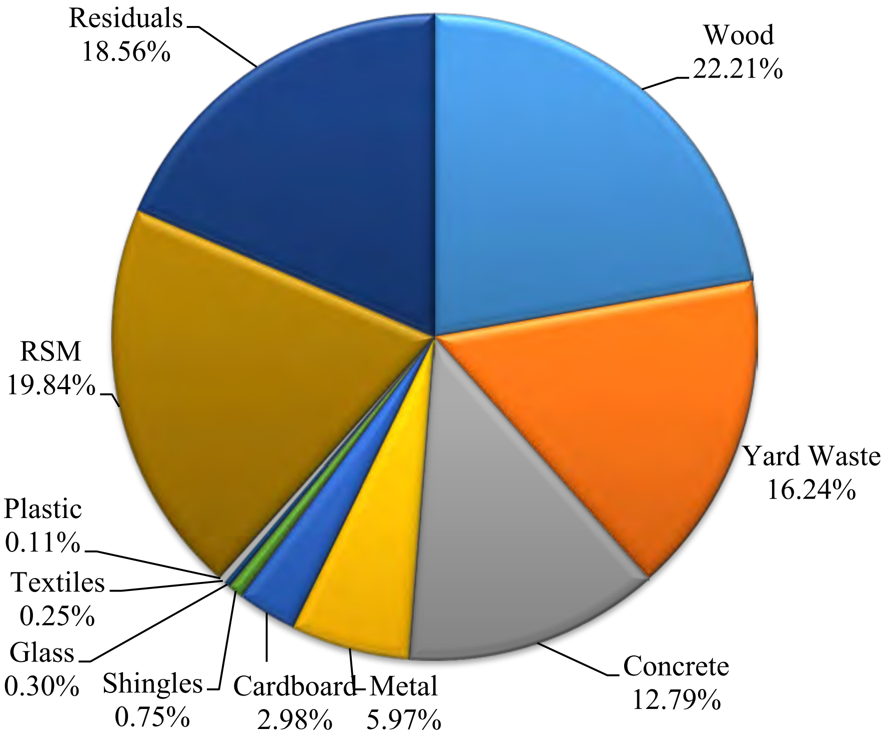

Figure 2-2. Weighted Average Mass Fraction of Materials Recovered or Discharged from Five Florida CDD Processing Facilities (developed from Calhoun (2012))

Table 2-8. Proposed LCI Dataset: Mixed CDD Processing Facility

Input Flow	**Source**	**Category**	**Unit**	**Amount**
Mixed C&D		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Diesel, combusted in industrial equipment	Calhoun (2012)	Flows	L	0.00199
electricity, at industrial user		Flows	kWh	0.0250
Output Flow	**Source**	**Category**	**Unit**	**Amount**

Wood, from CDD processing facility	*Calhoun (2012)*	*Construction and Demolition Debris Management*	*kg*	*0.222*
Yard waste, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.162
Concrete, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.128
Metal, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.0597
Cardboard, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.0298
Shingles, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.00752
Glass, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.00299
Textiles, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.00251
Plastic, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.0011
RSM, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.198
Residuals, from CDD processing facility	Calhoun (2012)	Construction and Demolition Debris Management	kg	0.186

2.5.8 Landfilling

2.5.8.1 Background

Although recycling rates have risen steadily based on increased consumer awareness, grants, and other incentives to encourage recycling, landfilling is still the predominant EOL management of discarded materials in the US, primarily attributed to reduced cost. While CDD disposal typically occurs at CDD or inert debris landfills, some CDD materials are disposed of at MSW landfills at their EOL phase. US EPA (2014) estimated that the average CDD mass fraction of incoming loads received at US MSW landfills was 10.5% based on the results of 12 large-scale waste characterization studies.

The materials disposal LCI should include materials and energy inputs and emissions associated with landfill construction, waste placement and compaction, and closure and post-closure-care activities along with the long-term liquids and gaseous emissions pertaining to biogeochemical decomposition of deposited materials. The domain of disposal related inputs and outputs considered varies significantly among different LCA models and LCI databases. For example, US EPA’s WARM only consider GHG emission from materials transport, waste placement and compaction activities, whereas WRATE includes materials and inputs, and emissions associated with landfill construction, operation, closure. Several other LCA models account for only a smaller subset of these emissions. This section presents LCI for landfill construction, operation, closure and post-closure activities.

2.5.8.2 Construction

Landfill construction requires a variety of material and energy inputs. Landfills are built as containment systems with the goal of minimizing direct (e.g., waste-related) emissions to the surrounding environment. The bottom liner of landfill cells is generally constructed via a combination of low-permeability (typically $<10^{-7}$ cm/sec) compacted earthen material and geosynthetic materials (typically 60-mil-thick HDPE). The purpose of the bottom liner is to contain and collect leachate and remove it out of the landfill cell, generally using porous drainage media (e.g., gravel), piping, and mechanical pumps to prevent build-up of liquid on the liner system. Emissions resulting from landfill cell construction occur during liner material manufacturing, transport and use of heavy equipment for on-site soil excavation and liner installation.

The fuel consumption and material resources required for landfill construction would depend on the level of environmental controls installed at the site. Composite liner systems are frequently installed at MSW landfills, which often include multiple layers of different geosynthetic materials for both leachate-collection and leak-detection purposes. Geosynthetics may be placed in contact with underlying low-permeability earthen material, which commonly will require compaction prior to the placement of the geosynthetics. In addition to the energy consumption of equipment needed to transport, place, compact, and weld liner components, the environmental burdens resulting from geosynthetic manufacturing should be taken into account.

Ecobalance developed an LCI for US MSW landfills for the Environmental Research and Education Foundation in 1999 based on a survey of more than 100 MSW landfills across the US and Europe. Part of this survey included compiling the average characteristics and fuel demand necessary for the construction of the liner system and other support infrastructure. Ecobalance (1999) provides the average thickness and the density of each of the liner components and, based on survey results, presents the average airspace use per MSW landfill footprint area. Ecobalance data were used as a primary input for developing landfill construction LCI for MSW-DST and EASETECH.

The upper end of the literature-reported density range of CDD materials as reported by Jambeck (2004), 359 kg/m^3, was used in this analysis to estimate the MSW landfill footprint required per mass of CDD material accepted at MSW landfills. The upper end of the range was selected as these density values are, probably, representative of uncompacted CDD; in-situ density of CDD placed in landfill is expected to greater than uncompacted CDD. This allowed an estimate of the mass of individual materials needed per mass of CDD material accepted at an average MSW landfill. Ecobalance (1999) also developed an estimate for the amount of diesel fuel required for MSW landfill construction on a mass-acceptance basis. The report details the average transport distances for each construction material; these distances were multiplied by their respective masses and the resulting mass-distance amounts were organized and summed by whether materials were transported more or less than 35 km. Material transport greater than 35 kilometers was modeled as long-haul transport and all other transport was modeled as short-haul.

Ecobalance (1999) presented an estimate the quantity of steel required to manufacture the equipment used for construction. This estimate was performed by assuming the average lifespan and weight of a wheeled scraper as 17,000 operational hours and 49,837 kg, respectively. The equipment hours used per mass of waste was divided by the total lifespan operation hours of the equipment and then multiplied by the total weight of the equipment.

Table 2-9 presents the proposed LCI dataset for MSW landfill material and energy construction burdens associated with the placement of CDD materials at an MSW landfill site. CDD landfills, typically, are not lined as no federal requirements for liners and leachate collection systems exist for CDD landfills; some states require liner construction for CDD landfills. Data that detail the energy and material burdens associated with the construction of an unlined CDD or inert debris landfill were not found. Golder Associates (2005) used MSW landfill construction LCI as a proxy for CDD landfill for WRATE model due to lack of data. No construction LCI are developed for unlined CDD landfill construction due to lack of data.

Table 2-9. Proposed LCI Dataset: MSW Landfill Construction, for CDD Materials

Input Flow	Source	Category	Unit	Amount

Clay, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.15
HDPE Liner, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.00019
Geotextile, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	3.80E-05
Sand, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.063
HDPE, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	2.60E-06
Steel, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	1.60E-05
PVC, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	2.00E-06
Asphalt pavement, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	8.50E-05
Concrete, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	9.00E-05
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Ecobalance (1999)		t*km	0.0032
Truck transport, class 8, heavy heavy-duty (HHD), diesel, long-haul, load factor 0.75	Ecobalance (1999)		t*km	0.00011
Diesel, combusted in industrial equipment	Ecobalance (1999)	Flows	L	0.00022
Output Flow	**Source**	**Category**	**Unit**	**Amount**
MSW landfill construction, for CDD materials		*Construction and Demolition Debris Management*	*kg*	*1*

2.5.8.3 Operations

Waste placement and compaction begins following construction of the liner/leachate collection system. Waste is filled in designated cells and lifts in a sequenced filling plan. Landfill operations generally include placing and compacting waste materials as well as periodically applying cover soil to the exposed waste surface. MSW landfills will commonly install a daily cover over the active waste face while CDD sites may install a weekly cover or no cover at all. Besides the diesel energy necessary to place and compact incoming waste, electricity is necessary to power numerous site facilities and buildings (e.g., scalehouse, workshop, offices, lighting). Cover soil is assumed to represent 10% of the volume of waste material placed at MSW landfills. However, cover soil is assumed to only represent 1.43% of the volume of waste material placed at CDD landfills (i.e., it is only placed once on a weekly basis or 1/7th of the daily cover amount). Literature-reported CDD material densities were used to estimate the corresponding mass of cover soil necessary for material placement at either a CDD or MSW landfill site, assuming a soil density of 1,330 kg/m^3 (USDA 2013). These densities were necessary in order to translate cover soil requirements from a volumetric to a gravimetric basis.

Ecobalance (1999) provides operational diesel requirements with and without daily cover soil application. However, no estimate of electricity consumption is provided. IWCS (2014b) compiled and analyzed electricity consumption and waste acceptance data from a regional MSW landfill in north-central Florida to estimate electricity consumption on a per kg waste mass basis. All electricity use was included in the estimate for MSW landfills except for electricity required for the recycling center, leachate pumping (e.g. sumps), and leachate treatment (e.g. leachate aeration ponds). Leachate collection and treatment will be handled in a separate LCI process dataset. Electricity consumption at a CDD landfill was estimated based on IWCS (2014b) data, but only included the electricity demand from the site's office and workshop. The nationwide average emissions from waste collection and transport to each landfill site category were not found; however, a transport distance of 20 km was assumed and included in each CDD material's disposal datasets. Table 2-10 and

Table 2-11 present LCI datasets that estimate energy consumption during the operation of CDD and MSW landfills, respectively. CDD material-specific cover soil requirements depend on the density of the CDD material and are therefore included in the material-specific disposal process datasets presented in the subsequent chapters.

Ecobalance (1999) presents an estimate of the quantity of steel used for equipment used to place and compact each unit mass of waste according to whether the site has daily or weekly/no cover soil requirements. It was assumed that landfills with a daily cover soil requirement are reflective of MSW landfill practices while sites with a weekly/no cover soil requirement would be representative of CDD landfill practices. Steel requirement estimates were performed by assuming the average lifespan and weight of a refuse compactor as 8,000 operational hours and 32,821 kg, respectively. The equipment hours used per mass of waste was divided by the total lifespan operation hours of the equipment and then multiplied by the total weight of the equipment.

It should be noted that these LCIs do not include emissions associated with operations equipment decommissioning or for manufacturing/disposing service and maintenance consumables (e.g., lubricating oil, rubber tires) due to lack of US-specific data. Golder Associates (2005) compiled material usage for WRATE; the sources of data used to develop these estimates were not provided by WRATE.

Table 2-10. Proposed LCI Dataset: CDD Landfill Operations

Input Flow	Source	Category	Unit	Amount
Diesel, combusted in industrial equipment	Ecobalance (1999)	Flows	L	0.00077
Electricity, at industrial user	IWCS (2014b)	Flows	kWh	0.00064
Steel, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.00011
Truck transport, class 8, heavy heavy-duty (HHD), diesel, long-haul, load factor 0.75	Ecobalance (1999)		t*km	4.4E-05
Output Flow	**Source**	**Category**	**Unit**	**Amount**
CDD landfill operations		*Construction and Demolition Debris Management*	*kg*	*1*

Table 2-11. Proposed LCI Dataset: MSW Landfill Operations

Input Flow	Source	Category	Unit	Amount
Diesel, combusted in industrial equipment	Ecobalance (1999)	Flows	L	0.0012
Electricity, at industrial user	IWCS (2014b)	Flows	kWh	0.0013
Steel, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.00016

Truck transport, class 8, heavy heavy-duty (HHD), diesel, long-haul, load factor 0.75	Ecobalance (1999)		t*km	6.4E-05
Output Flow	**Source**	**Category**	**Unit**	**Amount**
MSW landfill operations		*Construction and Demolition Debris Management*	*kg*	*1*

2.5.8.4 Closure and Post-Closure

Once a landfill has reached permitted capacity, it will undergo a closure process that usually involves installing a low-permeability or impervious final cover system to minimize precipitation infiltration and landfill gas emission. A gas collection and control system (GCCS) is typically installed at an MSW landfill before closure to control fugitive LFG emission. Unlike MSW landfills, CDD landfills do not typically have an active GCCS. LCI associated with closure include cap installation (material and energy usage), construction of other site infrastructure (such as roads), and continued operation of leachate and gas collection and management systems, environmental monitoring, and post-closure care activities.

Ecobalance (1999) summarizes the quantities of individual cap materials necessary to close an MSW landfill, based on a "typical final closure cover profile," which includes layers of soil, geotextile, sand, clay, and HDPE. Consumption of soil and clay materials is aggregated by Ecoblance (1999) into a single "soil" material category to quantify fuel consumption for soil production- and transport-related emissions. This material combination was preserved in the LCI dataset provided in Table 2-12. The materials necessary for installing a GCCS and gas monitoring system are also provided for the closure phase of the MSW landfill and are organized into the consumption of HDPE and PVC.

Ecobalance (1999) provides an estimate of the quantity of steel used for manufacturing of equipment steel used to place and compact each unit mass of waste by assuming the average lifespan and weight of a wheeled scraper as was assumed for landfill construction (i.e., 8,000 operational hours and 32,821 kg, respectively). The equipment hours used per mass of waste was divided by the total lifespan operation hours of the equipment and then multiplied by the total weight of the equipment.

During the post-closure-care period (assumed 30 years), Ecobalance (1999) assumes that 10% of the cap will need to be replaced due to erosive wear. The proposed LCI includes the soil and fuel (diesel) requirement for replacing 10% of the cap over the 30-year post-closure-care period. The soil and fuel amounts provided by Ecobalance (1999) for closure were increased by 10% to account for this additional soil needed over the post-closure care period. The fuel consumption resulting from site inspections (eight inspections were assumed to occur annually) and site mowing is also estimated on an annual basis. These emission factors were multiplied by the 30-year post-closure-care period and are included in Table 2-12 as "Gasoline combustion, in industrial equipment."

From an LCI perspective, constructing and operating a GCCS entails emissions from producing and transporting system components and energy demands from GCCS construction and installation. GCCS commonly include a flare or other destruction device (e.g., an internal combustion engine) to oxidize methane and other chemicals of concern to carbon dioxide. However, the LCI presented below do not include materials and energy input for constructing a blower/flare station.

Table 2-12. Proposed LCI Dataset: MSW Landfill Closure and Post-Closure, for CDD Materials

Input Flow	Source	Category	Unit	Amount
Soil, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.42
HDPE geomembrane, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.00012
Geotextile, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	3.90E-05
Sand, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	0.13
HDPE, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	8.20E-06
PVC, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	4.20E-06
Steel, at production	Ecobalance (1999)	Construction and Demolition Debris Management	kg	6.20E-06
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Ecobalance (1999)		t*km	0.0077
Truck transport, class 8, heavy heavy-duty (HHD), diesel, long-haul, load factor 0.75	Ecobalance (1999)		t*km	7.20E-05
Diesel, combusted in industrial equipment	Ecobalance (1999)	Flows	L	7.40E-05
Gasoline, combusted in industrial equipment	Ecobalance (1999)	Flows	L	6.20E-07
Output Flow	**Source**	**Category**	**Unit**	**Amount**
MSW landfill closure, for CDD materials		*Construction and Demolition Debris Management*	*kg*	*1*

2.5.9 Landfill Leachate Emissions

Landfill leachate is generated as precipitation or waste-entrained moisture percolates through the waste material and dissolves and retains various compounds. Unlined landfills or landfills with damaged bottom liners have the potential to release landfill leachate to underlying soils and groundwater. In LCI databases and models these are generally modeled as emissions to water, though leachate may contain dissolved gaseous species that are ultimately released to the atmosphere. As mentioned previously, CDD landfills do not carry a federal requirement for liners and leachate collection systems like MSW landfills. Leachate emissions are caused by the release of compounds/elements in the waste materials themselves, resulting in direct (i.e., waste-specific) emissions. Models and/or databases often do not handle leachate emissions on a waste-specific basis but rather on assumptions of leachate composition from mixed waste streams (e.g., MSW leachate), due to the relative lack of data on emissions from individual waste components, particularly over large spans of time. The timeframe over which leachate emission continues to occur after the waste placement complicates estimations of long-term leachate emissions. The handling of leachate emissions by different models is described below. It should be noted that most of these models are specific to MSW and, in general, do not include data for CDD materials.

2.5.9.1 WARM

WARM estimates the greenhouse gas (GHG) emissions associated with the EOL management of various materials, including CDD constituents. Based on a screening analysis by Freed et al. (2004), US EPA (2012a) recognized that an insignificant fraction (<1%) of the carbon input into a landfill dissolves into leachate. Leachate carbon storage was not ultimately included in the model; emissions of other contaminants such as heavy metals, anions, and cations are not considered as well. Leachate recirculation to enhance the waste decomposition rate is recognized and the model provides LFG-specific GHG emissions for four levels of moisture content (qualitatively specified as dry, average, wet, or bioreactor) (US EPA 2012a). The GHG emissions from electricity and fuel use associated with leachate collection and management are not included.

2.5.9.2 Athena's Building Impact Estimator

The Athena Impact Estimator for Buildings does not include emissions associated with leachate collection, treatment, or discharge of untreated/treated leachate into the environment.

2.5.9.3 WRATE

The leachate-specific emissions description presented in this section is primarily based on a report by Golder Associates (2005). The Waste and Resources Assessment Tool for the Environment (WRATE) uses material-specific leachate emissions for 16 primary waste categories typical of MSW. While the majority of the primary waste categories included in the model are specific to MSW, some CDD materials (e.g., wood, ferrous and non-ferrous metals) are also included (Golder Associates 2005, 2014). While the documentation discusses leachate emissions associated with inert wastes, this waste stream is not included in the current version of the model.

WRATE provides leachate emissions for three predetermined landfill sizes (2.25-, 5-, and 10-million MT capacities) and three liner types (engineered clay, HDPE/clay composite, and dense asphaltic concrete) and estimates emissions of several contaminants identified by Hall et al. (2001) using LandSim (version 2.5 was the most current version and in use by WRATE at the time of the publication of this report). LandSim is a probabilistic model developed by Golder Associates for the United Kingdom (UK) Environment Agency for modeling leachate emissions to groundwater using probability density functions for parameters such as the number of pinholes and tears in the landfill liner (Drury et al. 2003; Golder Associates 2005).

WRATE models landfill-related emissions for a 20,000-year period. LandSim accounts for physical and chemical deterioration of the flexible membrane liners over time and assumes that the leachate extraction and treatment would cease following the post-closure-care period. The model does not consider deterioration of mineral components (clay, geosynthetic clay liner, dense asphaltic clay liner) with time.

The total mass loading to the environment is the sum of the loading to groundwater through leakage in the bottom liner and mass loading to surface water following treatment at a leachate treatment plant; only leachate collected (remaining after leakage from the base of the site) is treated before being discharged into the environment. The mass loading rates are estimated based on temporally-varying contaminant concentrations, leachate collection/leakage rate, contaminant-specific leachate treatment efficiencies, and waste amounts.

Leachate-related emissions for 30 inorganic and organic contaminants are modeled for MSW. The initial contaminant concentrations are either based on LandSim default concentrations (primarily based on Robinson (1995)) and data reported by Knox et al. (2000) and Robinson et al. (2004). The data published by Robinson (1995) and Knox et al. (2000) could not be located for a detailed review. Robinson et al. (2004) presented composition of leachate from different type of facilities accepting different waste types (incinerator ash, processed MSW, untreated MSW). For example, the leachate quality data for a landfill accepting untreated MSW are based on limited numbers of samples collected from three landfill sites in Europe.

The emissions of non-volatile and volatile species are assumed to exponentially decline with respect to L:S ratio and time, respectively. The decline rates are contaminant specific and are based on a laboratory-scale leaching test of four 20-kg samples collected from a landfill site in North Lincolnshire, UK. The concentrations of various contaminants leached at multiple L:S ratio were measured.

LandSim estimates leachate generation based on effective rainfall, a parameter estimated using both precipitation and an infiltration rate depending on cap. Leachate generation for the first 20 years is assumed to occur instantaneously as the waste is exposed to the precipitation. The model assumes the presence of a polyethylene flexible membrane liner (FML) cap after 20 years and considers FML degradation over time for estimating the infiltration rate. The onset of degradation is assumed to begin in 250 years and the end point is assumed to occur 1,000 years after landfilling ceases (when grassland infiltration rates are assumed after the degradation endpoint).

Leachate leakage through the liner system is estimated assuming 1 m of head on the liner, the number of tears and holes as well as liner oxidation, hydraulic conductivity and thickness of compacted clay or DCA, and surface area of the liner (the regulatory limit for the UK). The performance of DCA, geosynthetic clay liners and compacted clay is assumed to remain constant over time. HDPE liners are assumed to degrade over time, similar to the landfill cap, where the area of defects doubles periodically (default time of 100 years) (Drury et al. 2003). Leachate emissions to groundwater are based on contaminant transport modeling through the mineral part of the liner system and the unsaturated zone above the groundwater table.

WRATE assumes that collected leachate is pumped from the landfill to a leachate treatment plant, though no emissions associated with electricity or fuel usage for leachate conveyance are included in the model (Golder Associates 2005). Contaminant-specific treatment efficiencies are based on research by Robinson and Knox (2004) and a (unreferenced) personal communication by Robinson (2004). The treatment efficiencies range from 0% (e.g., Cl-, K+, Na+) to 95.5% (NH_3-N). The efficiency data for chemically similar compounds or elements used for contaminant treatment efficiency were not available. The total loading for each contaminant was allocated to individual waste streams by an “expert panel” of 14 individuals comprised of academic researchers, operators, consultants and regulators. Individuals within the panel were allowed to present their thoughts to the group for those waste streams for which they had specific knowledge. Additional details on the allocation process were not available. Although the documentation discusses the estimation of a loading factor based on elemental content of various materials, the role of this factor on the leachate emissions estimate is not clear. Moreover, it is not clear whether loading factors include emissions associated with leachate treatment plant residuals management.

2.5.9.4 MSW-DST

MSW-DST leachate LCI datasets include the release of effluents from leachate treatment and the release of uncollected leachate to the environment. The LCI also includes energy and materials required to transport and treat leachate. The model calculates environmental contaminant loading rates based on an estimated leachate generation rate, leachate collection efficiency and treatment efficiency, and contaminant concentration for each waste constituent for three landfill types: traditional, bioreactor, and ash.

The leachate generation rate is estimated based on precipitation and a time-varying precipitation fraction that enters the landfill. The model assume 20%, 6.6%, 6.5%, and 0.04% of the total precipitation enters the landfill from 0-1.5, 1.5-5, 5-10, and after 10 years of waste placement, respectively, based on field data reported for multiple sites (Ecobalance 1999). Landfill documentation by NCSU and ERG (2011) is not clear whether the same approach and time-varying precipitation fractions are used for estimating leachate generation for all three landfill types. Also, it is not clear whether the model uses region-specific precipitation for the leachate generation rate (selected from a total of nine US regions). The model assumes a leachate collection and treatment time horizon of 100 years. The model assumes an insignificant leachate generation in the post-closure period (after 100 years) due to the placement of a low-permeability cap at the end of the operating period.

Some discrepancies were identified between the model documentation (NCSU and ERG 2011; Sich and Barlaz 2000) and the current version of the software, which has been modified since publication of the model documentation. For example, model documentation states that the timeframes for leachate collection and treatment can be user-specified, however, the current version of the model does not offer this flexibility.

A leachate collection efficiency of 99.8% is used to estimate the leachate collection rate. The model assumes that 100% of the collected leachate is sent to a treatment plant for the entire 100 years for all three types of landfills except for the first 20 years for bioreactor landfills. The leachate collected from a bioreactor landfill over the first 20 years is assumed to be recirculated. The model considers biochemical oxygen demand (BOD), chemical oxygen demand (COD), ammonia (NH_3), phosphate (PO_4), total suspended solids (TSS), arsenic, cadmium, chromium, lead, mercury, selenium, and silver emissions for MSW landfills. The model documentation also lists several hydrocarbons, but these do not appear to be included in the model. The model considers COD, NH_3, PO_4, arsenic, cadmium, chromium, copper, iron, lead, mercury, selenium, silver, and zinc for leachate from ash monofills. A constant concentration throughout the modeling period is assumed for ash monofill leachate.

The concentrations of all the contaminants except BOD and COD were assumed to be constant over time. The BOD concentration was assumed to be 10,000 mg/L for the first 1.5 years (1 year for bioreactor landfills), linearly declining from 10,000 mg/L to 1,000 mg/L from year 1.5 to year 10 (from year 1 to year 3 for bioreactor landfills), linearly decreasing from 1,000 mg/L to 10 mg/L from year 10 to year 50 (year 3 to year 10 year for bioreactor landfills), and a constant concentration of 10 mg/L after 50 years (10 years for bioreactor landfills). The COD concentration is estimated based on BOD concentration and the BOD/COD ratio is adjusted based on waste age. A COD concentration of 12,500 mg/L for the first 1.5 years (1 year for bioreactor landfills), linearly declining from 12,500 mg/L to 3,333 mg/L from year 1.5 to year 10 (year 1 to year 3 for bioreactor landfills), linearly decreasing from 3,333 mg/L to 1,000 mg/L from year 10 to year 50 (year 3 to year 10 for bioreactor landfills), and a constant concentration of 1,000 mg/L after 50 years (10 years for bioreactor landfills).

TSS, NH_3, PO_4, and metal concentrations in leachate for MSW landfills are reported as based on data reported by Ecobalance (1999). The ash monofill leachate contaminant concentration data are based on the leachate quality data from five ash monofills reported by US EPA (1990). US EPA (1990) reported ash characterization data based on five individual composite ash samples collected from five incineration facilities and one to seven leachate samples were collected from each of the landfill sites that accepted ash from these facilities (where all but one of the landfill sites were ash monofills).

LCI emissions related to leachate transport to a wastewater treatment plant (WWTP) are based on travel distance, leachate load, and the pre-combustion and combustion emissions of fuel used for transport. However, the information source for these emissions was not available in the model documentation. The emissions from treated leachate discharge are estimated based on treatment efficiencies in Robinson and Knox (2003), US EPA (1989), and US EPA (1992) (as cited in NCSU and ERG 2011) for an average WWTP. Treatment efficiencies range from 21.6 to 98% removal depending on the constituent- e.g., 21.6% treatment removal for PO_4 and 98% for NH_3.

The model also includes emissions associated with electricity use for leachate treatment and biogenic carbon dioxide emissions associated with BOD removal. A user-specified regional energy mix can be used for energy-consumption-related emissions (Dumas 1999). The model also estimates sludge production associated with BOD, PO_4, metals, and TSS removal.

The model calculates the total amount of these contaminants released into the environment (referred to as a *contaminant yield)* (mass of contaminant per unit waste mass) based on the model's leachate generation and collection rates and post-treatment contaminant concentrations. The generic MSW contaminant yields are allocated to different waste materials. The BOD, COD, and TSS yields are allocated based on the LFG attributed to the waste components. NH_3 and PO_4 are allocated to waste fractions based on the initial concentration of these contaminants for different waste materials reported by Barlaz (1997). Grass and food waste account for most of the NH_3 and PO_4. The generic MSW metal yields are allocated to individual waste components based on the total metal content of specific waste components, as reported by AJ Chandler & Associates Ltd. (1993) (as cited in NCSU and ERG 2011). A copy of the report by AJ Chandler & Associates Ltd. could not be found through an extensive web search for further review. Minor organic compounds, which are all hydrocarbons (11 in total), were reported to be considered in the model by older MSW-DST documentation authored by Sich and Barlaz (2000), but these trace organics are not included in the recent version of MSW-DST model documentation by NCSU and ERG (2011).

The emissions associated with electricity and materials (pipes, gravel, geotextile, pumps etc.) used for the leachate collection system are not accounted for in the MSW-DST. Electricity used by leachate pumps and ancillary equipment for the operational phase is not considered. The model calculates the amount of fuel, PVC, and concrete (i.e., consumables) used in the leachate recirculation system (LRS) (for bioreactor landfills) because the LRS is installed during waste filling. After the construction-phase, LCI financial and environmental parameters associated with operational consumables are considered as part of the LCI for the landfill. Horizontal trenches are assumed to be constructed with perforated PVC pipe and sand and the vertical wells are assumed to be constructed with perforated concrete pipe filled with gravel. The model does not account for sand and gravel used for constructing leachate recirculation devices. The model includes emissions associated with the production of fuel, PVC, and concrete used for bioreactor landfills.

2.5.9.5 EASETECH

EASETECH estimates leachate generation based on user-specified site geometry, net infiltration, and leachate collection efficiency that may be set differently for different time periods. EASETECH models the release of 33 contaminants with leachate and assumes a default leachate composition for four time horizons (0-2 years, 3-10 years, 11-40 years, and 41-100 years), which is independent of waste composition. The concentrations of the leachate constituents decrease with each subsequent time period. The model allows the user to assign unique leachate constituent concentrations for the different time periods.

Based on user-specified percentages, the model allocates leachate into collected and uncollected categories. The uncollected leachate is assumed to be discharged to surface water. EASETECH allows the user to simulate the quality/presence of a liner by specifying leachate collection efficiencies, which also may be changed for different time periods. The model accounts for the discharge of the 33 constituents released with either uncollected or with collected and treated leachate, where all the contaminants except COD are allocated to the surface water compartment; COD is allocated to an unspecified air compartment as fossil carbon dioxide.

Although the model does not include emissions associated with collecting and transporting leachate to a WWTP, a leachate transport process can be readily inserted into the model. The model accounts for energy and the associated emissions for leachate treatment; the model assumes 9 kWh of electricity consumption for the treatment of each 1,000 kg leachate. The total direct emissions associated with leachate release to the environment are estimated by summing the emissions from uncollected leachate with emissions from treatment plant discharged effluent; the contaminant loading within the effluent are based on user-specified treatment efficiencies. The model's default leachate composition is based on the data reported by Reinhart and Grosh (1998) and Kjeldsen and Christophersen (2001). Reinhart and Grosh (1998) reviewed Florida Department of Environmental Protection (FDEP) leachate composition data from 39 lined Florida MSW landfills to characterize Florida MSW leachate. Kjeldsen and Christophersen (2001) characterized leachate at 106 old MSW landfills and at one modern, lined Danish MSW landfill. Leachate samples from the old landfill sites were taken from either within the waste mass or within 50 meters of the landfill border. The timespan over which leachate samples were taken was not provided for these studies. The model cites the use of the data reported by Reinhart and Grosh (1998) for developing the pollutants discharged from the wastewater treatment plant, but no such data were found in Reinhart and Grosh (1998).

2.5.9.6 GaBi

Twelve of the 59 datasets identified to include landfill leachate are specific to the US in GaBi. Five of these landfill process datasets are for landfilling multi-component (i.e., heterogeneous) waste streams: biodegradable waste and MSW (under average, arid, moderate, and wet climatic conditions, for four total MSW datasets). Seven US landfill disposal datasets are related to specific wastes: ferrous metals, glass/inert waste, paper waste, plastic waste, textiles, untreated wood, and wood products (e.g., oriented strand board, particleboard). Landfill leachate parameters and processes described in the metadata for each US process dataset were identical and are described in this section.

Leachate-related emissions for the US process datasets consider leachate generation, collection, recirculation, and treatment. Leachate generation is determined based on precipitation. The US annual average precipitation is assumed for the process datasets except those specific to arid, moderate, and wet climatic zones. For climate-specific process datasets, the US states within those zones are identified and the zone-specific annual average precipitation is used. The US precipitation data were reported in the metadata to originate from the National Oceanic and Atmospheric Administration's (NOAA's) climatic data center.

The GaBi databases for the US are developed to be representative of national/regional data and the amount of waste landfilled assumed is based on research from US EPA and Levis and Barlaz (2011). Landfill characteristics are based on US national averages, attributed to Ham et al. (1999) and Themelis and Ulloa (2005). Themelis and Ulloa (2005) report quantities of waste landfilled (for 42 US states), operational US landfills, mean waste depth, density, and other characteristics.

The leachate collection efficiency in the US process datasets was reported in the metadata as 70% (though no reference was cited for this information). The leachate recirculation rate (i.e., % of leachate generated which is recirculated) was based on Benson (2007), where five field bioreactor sites that were examined showed 68.4% of leachate recovered and recirculated. Metadata states that 80% of US Landfills have some recirculation program (attributed to personal communication with Craig Benson (2012)). The product of these rates gives the total recirculation rate of 54.7%, which is used in the US landfill process datasets. Benson (2007) selected sites representative of contemporary US bioreactor practices, and conventional landfills were not examined.

The only leachate constituents US GaBi process datasets appear to include are COD and BOD. Metadata cites Kjeldsen et al. (2002) as the source for the COD/BOD ratio in MSW leachate (0.1). However, Kjeldsen et al. (2002) reports the long-term BOD/COD ratio in MSW leachate as 0.1, a ratio consistent with other sources. Kjeldsen et al. (2002) reports leachate data for many other constituents along with temporal variation characteristics; it is unclear whether any of this additional data are included in the GaBi datasets and if BOD/COD emissions are waste-specific.

Leachate treatment for US landfill process datasets occurs via activated carbon adsorption and flocculation (unreferenced in the metadata). Leftover sludge from the leachate treatment process is disposed of in the landfill and is assumed to be dried through combustion of natural gas. No information on the time period considered by US process datasets is included. There are other landfill process datasets containing leachate processes for areas outside the US; 47 landfill process datasets were found in GaBi for areas outside the US (typically countries within the EU). Similar to the US data, it is not clear whether leachate emissions are waste-material specific.

Non-US MSW landfill datasets include 60% transpiration/runoff of precipitation, no leachate recirculation, and 70% leachate collection efficiency (all unreferenced). Finnveden (n.d.) is cited in the metadata for assumptions of exponential solubility of fluids used for leachate-related solubility calculations. All non-US datasets use a landfill with fixed dimensions for a 100-year period. Leachate treatment includes activated carbon and flocculation with sludge disposal in a landfill.

Non-US CDD process datasets that include a “landfill leachate” process are as follows: construction rubble on inert matter landfill, landfill for inert matter (construction waste), and landfill for inert matter (with separate datasets for individual CDD components such as aluminum, steel, glass, glass/inert waste). The leachate technology descriptions for the non-US inert matter landfill process datasets match those of non-US MSW landfill process datasets, with changes to the transpiration rate and leachate collection efficiency to 50% and 60%, respectively.

2.5.9.7 Ecoinvent

Ecoinvent includes LCI for four types of landfills: inert, residual, slag, and sanitary (i.e., MSW). CDD materials are disposed of only in inert and sanitary landfills (Doka 2003a). Ecoinvent does not consider the environmental effect of leachate from inert landfills and residual and slag compartment landfill units do not accept CDD materials (e.g., they only accept ash, desulfurization residues, and industrial waste). Therefore, only sanitary landfill LCI dataset information is discussed in this section. Materials-specific emissions of 41 chemical elements are included for various waste constituents. The following waste constituents can be used as proxy for CDD materials: wood, wood ash, gypsum, and cardboard.

Leachate-related emissions are divided into short- (<100 yr) and long-term (year 100 to year 60,000) emission periods. Doka (2003a) addresses this temporal distinction as a somewhat arbitrary choice, inclusive of the time periods studied by Zimmermann et al. (1996) (as cited in Doka 2003a). The Ecoinvent model assumes that all the leachate produced over the short-term period is collected and treated at a WWTP and that the liner and leachate collection system remain intact throughout that period (Doka 2003a). At the WWTP, treatment sludges are incinerated at a municipal incineration facility and the incineration residuals are placed in a residual material or slag compartment landfill. After treatment at the WWTP, the effluent is modeled as discharged to surface water (Doka 2003c). Long-term leachate management assumes that after 100 years the collection system fails and all leachate produced is released to groundwater (Doka 2003a).

Theoretical emission potential of various elements is estimated based on the material-specific degradability rate and elemental content along with waste composition. *Degradability* refers to decomposition and mineralization of materials in landfill. Material-specific degradability is calculated based on the fraction of carbon released from individual materials during the first 100 years, based on data presented by Micales and Skog (1997). The degradability rate for cardboard, wood, and gypsum is estimated to be 32%, <3.3%, and 100%, respectively. In other word, 100% of gypsum is estimated to mineralize within 100 years. It is assumed that all the elements release from material with the same rate. For example, 100% of both calcium and sulfate in gypsum will mineralize. Some fraction of the elements once released from the waste matrix may undergo chemical transformation and be retained in the landfill, while the rest exit the landfill with gas or leachate. For example, a portion of sulfate (from gypsum) is reduced to hydrogen sulfide and leaves the landfill as a gaseous emission, another fraction of the sulfate may transform to metal sulfide and precipitate out, and the rest would be released with leachate. The amount of elements released with gas and leachate are estimated based on release factors, which are specific to elements.

An average release factor representing the expected release after taking into account elemental re-precipitation within the landfill for the different elements is calculated as the ratio of the actual to the theoretical leachate emissions of the element. The "actual" leachate emissions are estimated as the product of the leachate volume released over the first 100 years and the average leachate concentration of that element (assumed constant over the first 100 years) as estimated from eight studies analyzing MSW landfill leachate.

Total emissions for each element are estimated based on the degradability rate, the elemental concentration, release rate, and the fractional representation in MSW for each specific waste material. This estimate is partitioned into gaseous and leachate emission using % gas factors reported by Belevi and Baccini (1989). The elemental concentration of the different contaminants is assumed to be constant for the first 100 years.

Long-term leachate emissions in sanitary landfills are divided into two time periods; the first occurs following year 100 to year 4,500 and the second occurs following year 4,500 to year 60,000. A first-order decay model is used to estimate the long-term decrease in elemental concentration in leachate and to estimate the long-term leachate emission; as described above, Ecoinvent LCIs assume 100% of leachate over long-term is emitted into the environment without treatment.

Some emissions related to the construction of a leachate control infrastructure are considered by Ecoinvent. Sanitary landfills are modeled at a set size with a liner consisting of gravel, bituminous concrete, and polyethylene, and the energy demand for the liner as a whole is approximated as 0.5 L/m^2 of sealed surface (Zimmermann et al. 1996, as cited in Doka 2003). Eight concrete leachate tanks connected to the sewer are assumed. Consumables (e.g., PVC, cast iron, diesel fuel use) associated with the construction of the tanks, sewer pipe, and leachate collection pipes are inventoried in Ecoinvent. Doka (2003a) provides the LCI data (e.g., energy demand) of the infrastructure required for completing the Ecoinvent modeled sanitary landfill, which has a1.8-million-ton capacity.

2.5.9.8 Leachate Modeling Summary

Table 2-13 summarizes the leachate-related flows, which are considered in these models. An "X" indicates that the consideration of that flow is built into the model. A "P" indicates that only partial information was available for that flow. As shown in the table, only limited information is available on emissions to groundwater as a result of the leaching of waste materials. However, it should be noted that no information was found which estimated CDD material leachability in a CDD landfill or land-applied beneficial use application. Therefore, for the purposes of developing a landfill disposal management LCI for the different CDD materials presented in this report, literature was reviewed to identify sources of material-specific Synthetic Precipitation Leaching Procedure (SPLP) and Toxicity Characteristic Leaching Procedure (TCLP) data. SPLP data was selected as representative of leaching which occurs as a result of precipitation infiltration through CDD materials in either a CDD landfill or a beneficial use land application (e.g., use as a fill material). TCLP data was used to estimate leachability of CDD materials in an MSW landfill. The majority of the CDD materials were only modeled with respect to disposal in an unlined CDD landfill or land-applied beneficial use application. Due to an absence of information, the MSW landfill disposal LCIs of materials for which TCLP data were found (i.e. wood, RSM) do not include emissions information with respect to leachate collection, treatment and treated residual discharge; emissions to groundwater are presented as if no leachate collection/interception occurs.

Table 2-13. Leachate-related Inflow and Outflows in LCA Models

Flow	Construction Phase							
	EASETECH	**WARM**	**Ecoinvent**	**MSW-DST**	**WRATE**	**GaBi**	**ASMI IE Buildings**	**BEES**
Electricity (Pipe Welding)								
Emissions - LCS Installation (Piping, Gravel, Geotextile)								
Inputs - LCS Pipes and Fittings			X					
Inputs - Gravel/Geotextile			X					
Inputs - Leachate Pumps/Storage Tanks			X					
	Operation and Closure/Post-Closure Phase							
Electricity (Pumps and Ancillary Equipment)								
Electricity - Leachate Treatment	X			X		X		
Fuel - Leachate Transport to Treatment			P	X				
Emissions - Post-Treatment Discharge of Leachate	X		X	X	X	X		
Emissions - Leakage to Groundwater	X		X	X	X	X		

2.5.10 Landfill Gas Emissions

Landfill gas emissions result from the decay of landfilled organic materials and depend on the landfilled waste composition. In an anaerobic MSW landfill environment, LFG tends to be comprised of approximately 55% methane and 45% carbon dioxide with trace amounts of other gases for the majority of the landfill's active and post-closure life. The CDD stream typically contains smaller quantities of readily biodegradable wastes (generally the largest biodegradable component of CDD is wood and paper); thus a lower total bulk gas production is observed (Doka 2003a). However, at CDD landfill sites, H_2S, a malodorous compound produced typically from decay of sulfur-containing wastes (e.g., gypsum drywall), can be produced. Since gas-production rates are expected to be low at CDD landfills, there is no federal requirement for active GCCS at CDD, and employing combustion-based treatment systems can be challenging at CDD sites because of the small amount of gas produced.

Gas production at MSWs is typically assumed to follow a first-order decay relationship, and the production rates can be estimated using computer modeling tools (e.g., the US EPA's Landfill Gas Emissions Model) (US EPA 2005). Since a GCCS is rarely used at CDD landfill sites, the temporal modeling or estimation of gaseous emissions from CDD landfills (either controlled or uncontrolled) may be challenging because of the lack of data collected on gas composition and quantities at operating facilities. Several researchers have used field surface emissions monitoring and laboratory columns to quantify sulfur gas release from CDD landfills (Lee et al. 2006, Eun et al. 2007, Xu et al. 2014).

The generation, composition, and controlled/uncontrolled emissions of LFG are based on the gas-production properties of the landfilled material (e.g., decay rate, total gas production potential), GCCS operation and coverage area, and cover soil and cap installation and characteristics. Gas treatment or inhibition mechanisms (e.g., co-disposal of drywall with ash or lime) employed will also impact environmental emissions (Plaza et al. 2007, Panza and Belgiorno 2010, Xu et al. 2010, Sungthong and Reinhart 2011).

2.5.10.1 WARM

The Waste Reduction Model (WARM) estimates greenhouse gas (GHG) emissions associated with LFG from biodegradation of a variety of materials; dimensional lumber, wood flooring, medium-density fiberboard, yard trimmings (typical of land clearing debris), corrugated containers, and drywall are the CDD materials considered by the model to produce methane in a landfill. The model calculates material-specific emission factors for a variety of landfilling scenarios based on the measured initial carbon content, methane yield, LFG collection scenario, electricity generation from the collected and combusted methane, and carbon storage for 14 materials over 100 years. The initial biogenic carbon content and stored carbon contents used in the model are based on the material-specific data published by Barlaz (1998). Barlaz (1998) reported biogenic carbon contents and carbon storage factors of various materials in 2-L reactors in quadruplicate under anaerobic conditions. The carbon content of waste materials was measured at the beginning and end of the experiment to estimate the initial and stored carbon content, respectively (Barlaz 1998). The model uses data reported by Eleazer et al. (1997) to estimate the material-specific methane yield as a percent of the initial carbon content for various materials. Barlaz (1998) and Eleazer et al. (1997) used the data collected from the same set of experiments.

Adjustments to the methane yield reported by Eleazer et al. (1997) were made to account for 100% of the initial carbon if the methane and carbon dioxide yield and stored carbon did not add up to 100%; the volumetric carbon dioxide yield was not measured and was assumed to be equal to that of methane. For example, the methane yield for gypsum board was increased from 16% to 18% of the initial biogenic carbon content so that the methane and carbon dioxide yield, when added to the stored carbon content of 64%, accounted for 100% of the initial biogenic carbon. US EPA (2012a) identified proxies for the materials that were not included in the Barlaz (1998) study. For example, dimensional lumber and phone books were assumed to have the same characteristics as branches and newspaper, respectively. Water content of the different materials was adjusted as needed (US EPA 2012a).

The model provides emission estimates for three types of landfill operation: landfills without LFG recovery systems, landfills that flare LFG, and landfills that combust LFG for energy recovery. For the national average the model assumes that LFG is not collected for CDD landfills and that 28%, 38%, and 34% of the total methane generated is vented to the atmosphere, recovered and flared, and used for electricity generation, respectively, for landfills other than CDD landfills. CDD materials are assumed to be disposed of in a CDD landfill and therefore LFG collection from these materials is not considered by the model.

The fugitive methane emission is dependent on the time-varying methane generation rate and collection efficiency for the options with LFG collection and combustion. The model calculates LFG generation for four material-specific decay rates, using a first-order decay model. The material-specific decay rates reported by De la Cruz and Barlaz (2010) were used to estimate the LFG generation rate. De la Cruz and Barlaz (2010) used the data collected in the experiments reported by Eleazer et al. (1997) to estimate material specific-decay rates and compared the composite k value (calculated based on waste composition and measured material-specific k values) with the field k values (0.04 yr^{-1} and 0.12 yr^{-1}) to calculate a correction factor. The model uses the correction factor to adjust the lab-measured k-values to material-specific field-relevant decay rates of 0.02 yr^{-1}, 0.04 yr^{-1}, 0.08 yr^{-1}, and 0.12 yr^{-1} for dry, average, wet, and bioreactor landfill moisture conditions, respectively. Temporally varying collection efficiencies are applied to the generation rate to estimate the LFG collection rate for three LFG collection scenarios (typical, worst-case, and aggressive gas collection scenarios). However, the collection efficiency is the same for 6 to 100 years for all LFG collection scenarios (75% for the 6th and 7th year and 95% for the 8th - 100th). The LFG collection efficiency annually varies from 0 to 75% for the first 5 years and depends on the collection scenario. A temporally-averaged LFG collection efficiency, defined as the total methane collected over 100 years divided by the total methane produced over 100 years, is calculated for each material based on the total estimated LFG generation and annual collection rates.

The model assumes an oxidation rate of 10% for methane that is not collected by the LFG collection system. The oxidation rate is based on the data reported by Liptay et al. (1998) and Czepiel et al. (1996) and the IPCC (2006) in US EPA (2012a). Further investigation of these sources showed that this rate was based on work by Czepiel et al. (1996), who reported the methane oxidation rate based on methane flux measurements through laboratory-scale soil columns under varying temperature, soil moisture, and O_2 mixing ratios; soil samples were collected from a landfill cover in New Hampshire. Subsurface LFG migration is not mentioned by US EPA (2012a). The model also does not consider emissions of other gaseous contaminants.

The offset associated with LFG beneficial use is estimated only for electricity generation; offsets associated with other beneficial use applications such as direct use of LFG is not considered. The model calculates the offsets based on methane's energy content of 1,012 BTU per cubic feet, a methane density of 20 g/ft^3, an LFG-to-electricity heat rate of 11,700 BTU per kWh electricity, and a capacity factor of 85% (to account for system downtime). Regional (for nine regions in the US) utility mixes can either be user-specified or the US average utility emission factors can be used. The model provides material-specific offsets for non-baseload electricity generation. The model appears to assume 100% destruction efficiency for LFG combustion via flare and electricity generation. The model accounts for sequestering biogenic carbon, which is not degradable in an anaerobic environment. WARM does not account for emissions or materials consumed from manufacturing, installing, or operating the GCCS, energy recovery equipment, or LFG condensate-management devices.

2.5.10.2 Athena's Building Impact Estimator

Athena's Impact Estimator for Buildings (version 4.5, released in January 2014) includes an assessment of LFG produced from wood decomposition (Athena Institute 2013). LFG calculations are performed by the model to determine the biogenic carbon sequestration of wood during EOL management, where the amount of carbon in the product which is not released as LFG provides an environmental carbon storage benefit (Athena Institute 2013). The methodology for the biogenic carbon accounting was reported as based on the Publicly Available Specification (PAS) 2050 carbon footprint standard (BSI 2011). The model assumes 80% of waste for landfilling, 10% for recycling, and 10% for combustion. Emissions associated with LFG in the Impact Estimator for wood products are estimated based on LFG generation, LFG collection, uncontrolled LFG release, and the discharge of combusted LFG through flaring or energy recovery (though there is no accounting for energy offsets resulting from power production) (Athena Institute 2013).

The model uses a first-order decay model to estimate gas generation from anaerobic and aerobic landfills for 100 years. It is assumed that 23% of the wood decomposes to produce LFG. A decay rate of 0.04 year^{-1} is used for both aerobic and anaerobic landfill cases. LFG from aerobic landfilling is assumed to be constituted of only carbon dioxide, whereas LFG from anaerobic landfill is assumed to be constituted of 50% methane and 50% carbon dioxide by volume. The model assumes a collection efficiency of 82% and oxidation through cover material of 10%. The collected LFG is assumed to be combusted. The model does not offer credit for energy recovery from LFG combustion. The documentation does not provide sources of the inputs used for the LFG emission estimate and does not describe the methodology for estimating carbon stored from LFG emission estimates. It appears that the model does not consider emissions associated with GCCS equipment and material manufacture, installation, maintenance, or any consumables from GCCS operation.

2.5.10.3 EASETECH

EASETECH uses the Intergovernmental Panel on Climate Change's first-order decay model to estimate LFG generation (DTU 2013). EASETECH estimates total anaerobically degradable carbon for each waste material constituent based on its hydrogen, oxygen, nitrogen, and volatile solids content, as well as its methane generation potential (total methane volume produced per MT of volatile solids). The model calculates the methane fraction of the total gaseous emissions for each material constituent based on elemental composition (i.e. hydrogen, oxygen, nitrogen, and anaerobically degradable carbon). The model only allocates gaseous emissions to methane and carbon dioxide based on first-order decay of the anaerobically degradable carbon for each waste material fraction. The model adopts the first-order decay rates reported by De la Cruz and Barlaz (2010), who performed lab-scale decomposition studies to estimate material specific decay rates of 12 MSW components. Although similar, the decay rate constants do not exactly match those reported by De la Cruz and Barlaz (2010). While EASETECH provides decay rates for 22 material fractions, only eight unique decay rate constants are used in the model – similar material fractions were assigned the same rate constant (e.g. animal and vegetable food waste both have a constant of 0.137 year^{-1}). The model provides details on the chemical composition and amount of anaerobically degradable carbon within wood, concrete stones, cardboard, soil, and metal waste materials.

For each year that waste degradation is modeled, EASETECH calculates the total amount of anaerobically degradable carbon converted into LFG based on the material-specific first-order decay rates; the user may specify a different value for any or all of the material fractions. In addition to methane and carbon dioxide, 21 minor LFG constituents are added to the LFG on a g/m^3 concentration basis independent of waste type. The model allows concentration modifications or addition/subtraction of constituents as needed. While methane, carbon dioxide, and trace gas emissions will decrease over time according to the decay rate constants, EASETECH models the concentration of all gas constituents as independent of time.

The model includes a default 100-year period for the evaluation of LFG management, but the user may modify this time horizon. The model allows the user to specify LFG collection efficiencies for different time periods. The default collection efficiencies include 0% for the first two years, 80% for the next 43 years, and 0% for the remaining 55 years. The 20% of the LFG that is not collected from years 2-43 is further modeled according to "early" oxidation (i.e. the first 8 years during which LFG is collected) and late oxidation (i.e. the last 35 years during which LFG is collected) while the 100% of the LFG that is not collected for the last 55 years is modeled as undergoing late oxidation. A unique default oxidation/transformation rate is specified for 23 default organic and inorganic compounds, but these rates may be modified and additional substances may be added or subtracted. Methane is one example of a carbonaceous gas which is oxidized differently during the early/late oxidation periods – it is oxidized at 60% to carbon dioxide during the early phase and at 80% during the late phase.

The model does not account for subsurface gas migration process – LFG is either collected or released through the final cover. While documentation for LFG production, collection, and treatment within the EASETECH model is not explicit, it appears that Deipser et al. (1996) and Scheutz et al. (2004) were used to estimate the oxidation of methane and non-methane organic compounds (NMOC) in landfill cover soils for the different time periods. Scheutz et al. (2004) assessed the methane oxidizing potential of soil samples from a location bordering an unlined Danish landfill; offsite methane migration was observed at the sampling location. Soil samples from depths ranging from 0 to 90 cm were collected. Samples were placed in air-tight glass containers from which the air was evacuated and were subsequently incubated with gas mixtures consisting of 15% methane, 35% oxygen, and 50% nitrogen. Gas samples were routinely withdrawn from the containers after incubation for chemical analysis by gas chromatography to estimate oxidation of a variety of organic compounds.

EASETECH can model the collected portion of LFG as either utilized within an LFG-to-energy project or as sent to a flare for destruction. Power produced as a result of a gas-to-energy project is used to offset the power (and corresponding emissions) from a Danish coal-fired utility. The model estimates electricity generation from methane combustion using an estimated methane collection rate, an energy content of 37 MJ/m^3, and an efficiency of 30%. The model default gas-to-energy project also includes a heat-recovery process, which captures an additional 50% of the methane energy content. The model assumes a default energy recovery efficiency of 80% for combining heat and power processes. However, the user can change any of these efficiencies and remove either/both energy-recovery process(es). A methane and NMOC destruction efficiency of 97% is used for the flare and gas-to-energy processes. Again, while not explicitly stated within the model, it seems likely that flare emissions data from Frost et al. (1997) were used and potentially from NSCA (2002). The project team could not find a copy of these sources for additional data evaluation. The model does not account for emissions or materials associated with manufacturing, installing, or operating the GCCS, energy recovery equipment, or LFG condensate management devices (e.g., electricity used to operate gas mover equipment, pilot gas used for flare operation, lubricating oil needed for gas-to-electricity internal combustion generator sets). Only emission offsets associated with power production from energy recovery are included.

2.5.10.4 MSW-DST

The MSW-DST is focused on MSW materials; however, some LFG-generating MSW wastes in the model, such as cardboard and components of land-clearing debris (i.e., branches, grass, leaves), could be considered CDD materials. Gaseous emissions are not considered for ash landfills. The model estimates total LFG gas emissions and offsets for traditional and bioreactor landfills based on generation rate, collection efficiency, oxidation through landfill cover, electricity generation, and carbon sequestered (or stored). The LFG emission methodology used by the model is very similar to that used by WARM. The MSW-DST, however, offer more flexibility for user inputs. For example, DST allows users to specify the LFG collection efficiency for each year LFG is collected, where WARM does not offer such flexibility.

The model uses a material-specific methane generation potential and decay constant to estimate LFG generation for a user-specified MSW composition using a first-order decay model for a 100-year time frame. Similar to WARM, the material-specific decay rate and methane generation potentials used by the model are adopted from De la Cruz and Barlaz (2010) and Eleazer et al. (1997), respectively. LFG from anaerobic landfilling is assumed to be constituted of 50% methane and 50% carbon dioxide by volume. The model provides temporally-averaged default collection efficiencies for typical collection (national average) and state-of-the-art systems for traditional and bioreactor landfills. The model, however, allows users to select collection efficiency for each year from year 1 through year 100. The model does not account for landfill carbon sequestration – the justification given in the model documentation was that carbon storage was not included in other parts of the model outside of landfilling.

The model allows the user to select gas management methods for any year from year 1 through year 100. The management methods include venting to the atmosphere, combustion via flare, and combustion for energy recovery. The options for LFG combustion with energy recovery include combustion in internal combustion engines, turbines, and boilers. The energy generated by the LFG-to-energy system is calculated based on the volume of collected methane, conversion technology efficiency, and methane energy content (1,012 BTU/dry standard ft^3 methane). The emission offset LFG-to-energy emission offsets include precombustion and combustion emissions from fossil fuel use. The energy mix is then used by the model to determine the offset based on the emissions per kWh or MJ of the particular energy fuel in the energy mix. The fuels included in the energy mix are (with model default values shown in parentheses) coal (56.45%), natural gas (9.75%), residual oil (2.62%), distillate oil (0.23%), nuclear (22.13%), hydroelectric (8.59%), and wood (0.24%) (Dumas 1999).

The fugitive surface emissions are estimated based on the remainder of LFG, which is not collected by the GCCS or converted through oxidation in the cover soil. LFG soil migration or dissolution to groundwater is not considered in the MSW-DST landfill modeling process. The oxidation rate through the landfill cover can be specified as temporally varying (i.e., the user may specify different oxidation rates for each year of the entire 100-year time horizon).

The other trace constituents in LFG included by the model are benzene, chloroform, carbon tetrachloride, ethylene dichloride, methylene chloride, trichloroethene, tetrachloroethene, vinyl chloride, toluene, and xylenes; concentrations of these constituents are based on AP-42 values (US EPA 2008) and are modeled independently of MSW waste composition. Destruction of these organic compounds is generally modeled as >90% for all technologies (including flaring) and destruction efficiencies were based on research by Ecobalance (1999).

The MSW-DST includes emissions from LFG treatment/energy recovery, emissions which are solely based on the quantity of LFG combusted (NCSU and ERG 2011). Exhaust constituents emitted by LFG combustion/energy recovery equipment are as follows (reported in lb per dry ft^3 methane going through the equipment): carbon monoxide (CO), nitrogen dioxide (N_2O), particulate matter (PM), sulfur dioxide (SO_2), hydrogen chloride (HCl), and dioxins. These emission factors are also based on AP-42 values (US EPA 2008).

The gas collection and monitoring systems are assumed to be placed at landfill closure (NCSU and ERG 2011). The model calculates the materials necessary for installing a GCCS as well as the emissions from producing, transporting, and placing the materials by heavy equipment (e.g., fossil carbon dioxide released by fuel combustion in heavy equipment); the GCCS was assumed to be comprised of HDPE and PVC and installed at respective rates of 0.016 lb and 0.0081 lb per ton MSW for the gas collection system and 7.3×10^{-5} lb PVC per ton of MSW for the gas monitoring system. Rates of HDPE and PVC use for the gas collection and gas monitoring system were reported to be based on landfill site survey information gathered by Ecobalance (1999). The MSW-DST also estimates emissions from the final cover installation when refuse in cells reaches final grade (NCSU and ERG 2011).

The MSW-DST does not account for emissions from operating or maintaining the GCCS nor does it consider emissions associated with producing GCCS operation and maintenance consumables (e.g., lubricating oils, pilot gas). The model does not appear to consider emissions from electricity use by the gas collection system operation, emissions due to fuel use in the pilot light, or emissions associated with consumables used during flare station operation and maintenance activities.

2.5.10.5 WRATE

The Waste and Resources Assessment Tool for the Environment (WRATE) LCA model uses pre-developed output data from the software tool GasSim v2.0 to estimate LFG-related emissions for a series of landfilling scenarios. WRATE is primarily focused on MSW, though some materials which are commonly discarded in CDD debris also appear in the model as options for waste fractions and a modeling scenario could be run which focuses on specifically managing CDD debris. These materials include wood, combustibles (unspecified, carpet/underlay), non-combustibles (bricks, blocks, plaster), household hazardous waste (paint/varnish), ferrous metal, and several paper products. GasSim is a probabilistic performance assessment model that can be used to estimate gas (methane, carbon dioxide, and hydrogen) generation, partitioning between collection, migration, surface emissions, biological oxidation through cover, combustion plant, and atmospheric dispersion.

The rate of LFG generation is affected by the landfill size and waste composition (Golder Associates 2009). The emissions associated with the LFG collection are affected by the modeled collection efficiency and the different operational conditions and engineering practices such as LFG used for energy production, flaring, or lost due to surface emissions or subsurface migration (Golder Associates 2009). A key modeling parameter used in GasSimv2.0 is the "source term" for LFG (Golder Associates 2009). The source term for bulk gases is the LFG generation rate of that waste fraction (i.e., a total LFG yield) (Gregory et al. 1999); therefore, modifying the landfilled waste stream will affect the LFG generation rate (Golder Associates 2009).

In WRATE, there are three set landfill sizes based on filling rates of 125,000 MT/year, 250,000 MT/year, or 500,000 MT/year over a 20-year period. The period of gas collection is based on the LFG generation rate (i.e., there are threshold levels of LFG flow into the flare or LFG-to-energy which trigger their employment in the model), although LFG emissions are modeled over a 150-year period (Golder Associates 2009). Landfill gas generation follows a first-order decay model (deemed multi-phase, because of consideration of waste moisture content); thus it is time dependent (Gregory et al. 1999; Golder Associates 2009). The degradation rate (k) can be changed in WRATE and there are three options for decay rates. Decay rates are dependent on waste type and the moisture content of the landfill. The default WRATE setting is "normal" degradation, which combines slow, moderate, and rapid decay (due to the waste composition being a mixture of components, each of which has its own decay rate) (Golder Associates 2009). GasSimv2.0 includes lateral migration of LFG and trace constituents through the subsurface by advection and diffusion. Methane oxidation is assumed to occur through the landfill cap/cover at a rate of 10%, unchanging with time, and this estimate is based on a recommendation by the IPCC (2006). Specific information on the field or laboratory studies considered by IPCC (2006) for this estimate are not provided. No carbon storage is considered by the WRATE model (Golder Associates, n.d.). The model assumes a 20-year active lifespan of 10 cells, each filled in a 2-year period and progressively capped to optimize gas collection (Golder Associates 2009).

The primary gas constituents modeled by the first-order decay model are methane, carbon dioxide, and hydrogen (Golder Associates 2009). Total LFG production is equivalent to the sum of the hydrogen, carbon dioxide, and methane produced by landfilled waste. The generation of both bulk and trace LFG constituents is waste-material specific in GasSimv2.0. However, the gases are not simply allocated[1] to the waste fractions based on assumed generation rates; rather, bulk gas generation is estimated by GasSimv2.0 and a first-order decay function, based on the waste inputs and their characteristics (e.g., degradation rate), and detailed in Gregory et al. (1999), as reported by Golder Associates (n.d.). GasSimv2.0's bulk LFG calculation is based on work by Gregory et al. (1999), where waste fractions were grouped into waste which degrades slowly, at a moderate rate, and rapidly and by Barlaz et al. (1989) where cellulose, hemi-cellulose, lignin, and moisture contents of different waste streams are reported from a study which used 56 laboratory-scale lysimeters with shredded refuse to investigate decomposition (Gregory et al. 1999).

Golder Associates (n.d.) reports that trace gas emissions were based on multiple studies conducted from 1987 through 1997 (seven total studies), which examined VOCs in LFG at actual landfill sites. Trace gas constituents considered by WRATE are grouped into the following classes: alcohols, aldehydes, aliphatics, BTEX, CFCs, chlorinated solvents, chlorinated solvent degradation products, chloro-benzenes, HCFCs, hydrocarbons, partial combustion products, substituted aromatics, sulphurous compounds, and terpenes (for a total of 57 trace compounds, not counting isomers of these compounds). Waste-specific trace gas emissions were developed for GasSimv2.0 using a "top-down" approach. Concentrations of trace LFG constituents from research by Parker (2002) were assigned to the degradation of specific waste material

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[1] *Allocation* refers, in the case of WRATE, to taking the total amount of a constituent (e.g., CFCs) calculated or assumed and *then* using waste composition data to allocate what portion of that quantity is attributable to the given waste fractions.

fractions by an "expert panel" of industry professionals. This panel considered the amount of degradable carbon available in each of the waste materials when performing trace LFG constituent allocations.

Golder Associates (2009) developed linear equations relating gas emissions to the environment (kg/ton waste) to gas collection efficiency for each bulk and trace gas component. These linear equations, coupled with the reduction factors, were used to develop emission allocation tables in WRATE. WRATE users can edit these allocation tables, including changing the gas collection efficiency for a year when gas collection occurs (Golder Associates 2009).

LFG collection efficiency affects emissions, and there are six available gas collection efficiencies in WRATE: 0%, 30%, 65%, 75%, 90%, and 100%. These collection efficiencies only apply when gas is being collected and do not represent the total LFG collected over the landfill's lifetime. The fraction of LFG which is not collected is considered an uncontrolled release to the atmosphere through the landfill surface or through subsurface migration.

LFG treatment in WRATE can be specified as either occurring through combustion by flaring or in an LFG-to-energy project through the use of internal combustion engines (Golder Associates 2009). Because of the use of GasSimv2.0 to model a limited number of design scenarios, only a limited number of scenarios may in turn be modeled in WRATE (since the two programs are not coupled). For example, a WRATE user only has two modeling options for specifying a "flare-only" scenario: a 10-MT total landfill capacity, 30% collection efficiency, and slow degradation of biodegradable waste components; or a 5-MT total landfill capacity, 75% collection efficiency, and normal waste decay (Golder Associates 2009).

WRATE includes emission offsets resulting from an LFG-to-energy project by assuming a generating efficiency of 33% and 50 MJ/kg calorific value (for methane) (Golder Associates 2009). Discharge of treated LFG through an LFG-to-energy project (i.e., the "combustion plant") will partially destroy some pollutants and oxidize various LFG compounds.

WRATE accounts for raw materials (e.g., clay, concrete, steel) and fuel use during landfill construction, as detailed in Golder Associates (2005). Capping related materials and emissions are dealt with separately from construction-related emissions. Materials (e.g., cover soil) and fuel usage during landfill construction, operation, and closure are dependent on the landfill size selected. Golder Associates (2005) does not specify whether these emissions and materials for LFG construction and operation are related to the GCCS.

2.5.10.6It GaBi

LCI background documentation was reviewed for GaBi processes which model the disposal of various waste streams and which include LFG emissions data. US-specific GaBi datasets broadly include information on LFG generation, collection, uncontrolled release to the environment, and energy recovery. Several US-based landfilling processes for a variety of individual and mixed waste streams were identified for different climatic conditions (e.g., arid, moderate, wet).

These US-specific datasets rely on national average data for their LFG modeling parameters, however, regional precipitation information is mentioned within the process datasets (i.e., the ability to select arid, moderate, or wet climates). Landfill waste density, height, and area are attributed to Ham et al. (1999) and Themelis and Ulloa (2005). Themelis and Ulloa (2005) provide information on the quantities of waste landfilled for 42 states, the number of operational landfills, the area of the working face, average waste depth, density, and other parameters. Ham et al. (1999) was not available online for additional review. It is not clear from Gabi dataset documentation whether methane generation is waste-specific or time dependent, however, since the dataset considers the total methane generated and collected at US landfills (with data utilized from US EPA's Landfill Methane Outreach Program (US EPA 2013, LMOP 2013)), the temporal aspect may be included by assuming that the percent of landfill gas collected and managed by the process is the same as the nationwide average. A full citation for US EPA (2013) was not provided within GaBi process documentation; this information was unavailable for further review.

Numerous details regarding nationwide average LFG collection and combustion are summarized from US EPA (2013) and LMOP (2013) including the percent of methane collected for flaring, the percent collected for energy recovery, the percent released as fugitive emissions, LFG combustion efficiency and the fractions of energy recovered from combusted LFG available as electricity and thermal energy. LMOP (2013) information is based on field data representing 782 operational MSW landfill gas beneficial use projects. GaBi process documentation discusses the energy generated by LFG-to-energy systems at landfills, but the processes do not appear to include or consider the offsets associated with energy generation. Oxidation of methane in landfill covers was not discussed in any process datasets. All US datasets examined for LFG consideration are part of the GaBi Extension Database XVII: Full US (PE International 2013).

Emissions related to construction of a GCCS are not considered in US (or European) datasets, although the datasets do include emissions for producing and transporting materials to cover and line the landfill as well as fuels used for landfill operations (e.g., diesel emissions for compactors). No emissions associated with operation and maintenance consumables or energy use of the GCCS were included.

2.5.10.7 Ecoinvent

Doka (2003a) presented a methodology to estimate LFG generation, collection, uncontrolled release to the atmosphere, and emissions from LFG combustion. Model documentation specifically discusses the individual parameters of waste materials that uniquely influence their decomposition in a sanitary (i.e., MSW) landfill. Insufficient information was available in the documentation to provide a detailed discussion of gas modeling from non-MSW landfills.

Short-term (i.e., within the first 100 years) LFG generation is the only phase during which LFG emissions are modeled; air emissions after this period are considered negligible (Doka 2003a). Similar to the total emission of an element as leachate, the total emission of an element as LFG is dependent on the total fraction of the element in the waste stream of interest and the time- (i.e., short term, long term) and phase- (i.e., gas, liquid) specific transfer coefficient for that waste. Doka (2003a) developed waste-material-specific transfer coefficients for materials received at sanitary landfills, but uses an average transfer coefficient for wastes placed at inorganic landfills (e.g., residual and inert material landfills).

Transfer coefficients are estimated from an element's theoretical emissions potential, its release factor (estimated based on simplifications to leachability predictions which include consideration of element precipitation) and the fraction of the element emitted in LFG. The theoretical emissions potential is related to the carbon content of a given waste material; the carbon conversion and degradability rate for 13 different waste materials is provided in Doka (2003a) and carbon conversion rates are cited from Micales and Skog (1997). The fraction of the quantity of 19 different elements emitted in LFG is estimated from Belevi and Baccini (1989) (included in an appendix to Doka (2003a)).

Information on the quantity of LFG collected and combusted at a sanitary landfill was averaged from five literature sources reviewed and summarized in Zimmermann et al. (1996), however, this document was not available for review. In addition to methane and carbon dioxide (present at concentrations of 47% and 37% by volume), Doka (2003a) assumes that some amount of air intrudes into the waste mass and is collected along with LFG by the GCCS; oxygen and nitrogen are assumed to represent 2.5 and 13% of collected LFG (by volume). Besides carbon dioxide and nitrogen, combusted LFG is assumed to include quantities of carbon monoxide, non-methane volatile organic compounds, particulate matter (<2.5 microns), nitrogen dioxide and a relatively small amount of non-combusted methane. Emissions related to installation and operations of the GCCS, as well as consumables (e.g., fuel, oil) are not discussed. Subsurface gas migration and oxidation of methane through the landfill cover are not reported by Doka (2003a), however, hydrogen sulfide is assumed to partially oxidize in the top layer of the landfill and then appears to be inventoried completely as sulfur dioxide due to atmospheric oxidation.

The presence of hydrogen sulfide in LFG is assumed to be a result of gypsum disposal, where the concentration is dependent on the transfer coefficient for sulfur and the quantity of sulfur placed in the landfill. Gypsum decomposition for a 100-year period was assumed to be 100% (Doka 2003b). The sulfur transfer coefficient was estimated based on four hydrogen sulfide measurements from building waste landfill cells (Belevi and Baccini 1987) and an estimated 3% gypsum concentration in construction waste (Doka 2003b).

2.5.10.8 Landfill Gas Modeling Summary

Table 2-14 summarizes the LFG-related flows presented in the different LCA models. An “X” indicates that information with respect to that flow was identified in the model or in documentation for the model.

Table 2-14. Summary of Landfill Gas and GCCS LCI Information Included in LCA Models

Phase	Flow	EASETECH	WARM	Ecoinvent	MSW-DST	WRATE	GaBi	ASMI IE Buildings
Construction	Fuel (Well Installation/Pipe Placement)				X			
	Electricity (Pipe Welding)							
	Inputs - GCCS Pipes and Fittings				X			
	Inputs - Gravel/Geotextile							
	Inputs - Gas Mover and Ancillary Equipment							
	Emissions - GCCS Installation				X			
Operation and Closure/Post-Closure	Electricity (Gas Mover and Ancillary Equipment)			X				
	Fuel (Pilot Gas)							
	Inputs - O&M Consumables							
	Emissions - Gas-to-Energy Equipment	X	X	X	X	X	X	X
	Emissions - Gas-to-Energy Emissions Offset	X	X		X	X	X	
	Emissions - Flaring	X	X	X	X	X	X	X
	Emissions - Gas Cleanup/Treatment Equipment							
	Emissions - Surface Emissions/Venting	X	X	X	X	X	X	X
	Emissions - Cover Oxidation	X	X		X	X		X
	Emissions - Subsurface Migration/Dissolution into GW					X		
	Emissions - Carbon Storage Offset	X	X					X

The methodology used by the US EPA (2012b) fors WARM landfilling process was selected for estimating the gas emissions from the landfill disposal of CDD materials. US EPA (2012b) provides LFG production properties for numerous waste materials placed in landfills. While the specific focus of the documentation is MSW materials, several materials are also part of the CDD waste stream including dimensional lumber (selected to represent CDD wood), branches (selected to represent LCD), cardboard and gypsum drywall. The data presented in US EPA (2012b) used for estimating the LFG emissions for CDD and MSW landfill disposal of the different organic CDD materials discussed in this report is presented below in Table 2-15.

Table 2-15. Landfill Gas Production Properties for Different CDD Materials

Material	Initial % Carbon of Dry Mass (Barlaz 1998)	Methane Carbon as portion of initial carbon (%) (Barlaz 1998)	Ratio of Dry to Wet Mass	MSW Landfill Gas Collection Efficiency – Average Typical Landfill Scenario Moisture Conditions (Barlaz et al. 2009)
Dimensional Lumber/Branches[1]	0.49	0.12	0.9	0.9
Cardboard	0.47	0.22	0.95	0.89
Gypsum Drywall	0.05	0.18	0.94	0.87[2]

[1] US EPA (2012b) used the experimental gas production results from branches as a proxy for dimensional lumber. The Project Team uses branches gas production results as a proxy for LCD gas production in this report. A moisture content of 50% was used for LCD instead of 10% used by US EPA (2012b) for branches The methane and carbon dioxide emissions estimate for branches were adjusted for this moisture content difference.

[2] A gas collection efficiency for LFG produced from the decomposition of gypsum drywall was not provided, however, the gas collection efficiency for waste paper was assumed since this is the organic portion of the drywall which will produce methane/carbon dioxide.

The total amount of methane emitted from a landfill after the placement of any of the materials listed in Table 2-15 in a CDD landfill was calculated using the following equation:

$$M_{Gi} = C_{Di} \times M_{Ci} \times W_i \times \frac{16}{12}$$

Where,

M_{Gi} = methane generated from landfill disposal of 1 kilogram of the ith material (kg)
C_{Di} = initial carbon mass fraction of the dry ith material (%)
M_{Ci} = methane carbon as a fraction of the initial carbon (%)
W_i = ratio of dry to wet mass
16/12 = conversion factor methane to carbon mass

Assuming that LFG is not collected at CDD landfills, the total amount of methane emitted from CDD landfills was assumed to be 90% of the amount generated due to an assumed cover soil oxidation rate of 10% (as discussed previously in the WARM landfill gas emission section). The amount of degraded carbon is equally allocated to methane and carbon dioxide (based on results from Barlaz et al. 1989). Therefore, the amount of carbon dioxide generated from the placement of one of the CDD materials in a CDD landfill prior to cover soil oxidation may be approximated according to the following equation:

$$C_{Gi} = C_{Di} \times M_{Ci} \times W_i \times \frac{44}{12}$$

Where,

C_{Gi} = carbon dioxide generated from landfill disposal of 1 kilogram of the ith material (kg)
44/12 = conversion factor of the molecular ratio of carbon dioxide to carbon

The carbon dioxide emission estimated from equation above was multiplied with 1.1 to account for the carbon dioxide from the oxidation of methane in the cover soil.

US EPA (2012b) also presents information which allows an estimate of the amount of methane and carbon dioxide emitted from the MSW landfill disposal of the CDD materials presented in Table 2-15. US EPA (2011) reported that approximately 72% of all landfill-produced methane is generated at landfills with a GCCS. Methane emissions would result from fugitive LFG at MSW landfills without as well as with GCCS. The methane emissions from the disposal of the CDD materials at an average nationwide MSW landfill was estimated using the following equation:

$$M_{EMSWi} = M_{Gi} \times 0.9 \times \left(L_{GCCS} \times (1 - \eta_{Ci}) + (1 - L_{GCCS})\right)$$

Where,

M_{EMSWi} = mass of methane emitted from the disposal of 1 kilogram of the ith material at an MSW landfill (kg)
L_{GCCS} = percentage of methane from MSW landfills with a GCCS (i.e. 72%)
η_{Ci} = gas collection efficiency for the ith material (see Table 2-15)
0.9 = factor accounting for cover soil oxidation of uncollected methane. Other variables as defined above.

In addition to the carbon dioxide generated from organic material decomposition, carbon dioxide would also result from the combustion of methane collected in an MSW landfill GCCS and from the oxidation of uncollected methane emitted through the landfill cover soil. Carbon dioxide emissions from the disposal of the CDD materials at an average nationwide MSW landfill may be estimated according to the following equation:

$$C_{EMSWi} = C_{Gi} + M_{Gi} \times \frac{44}{16} \times \left((L_{GCCS}\eta_{Ci}) + 0.1 \times \left((L_{GCCS})(1 - \eta_{Ci}) + (1 - L_{GCCS})\right)\right)$$

Where,

C_{EMSWi} = mass of carbon dioxide emitted from the disposal of 1 kilogram of the ith material at an MSW landfill (kg)
0.1 = factor accounting for the carbon dioxide produced from cover soil oxidation of uncollected methane
44/16 = conversion factor of the molecular ratio of carbon dioxide to methane. Other variables as defined above.

Table 2-16 presents a summary of the estimated methane and carbon dioxide emissions for the CDD and MSW landfill disposal of the organic CDD materials discussed in this report based on the calculation methodology presented above.

Table 2-16. Methane and Carbon Dioxide Emissions for CDD and MSW Landfill Disposal of CDD Materials

	CDD Landfill Emissions		MSW Landfill Emissions	
Material	**Methane (kg/kg material)**	**Carbon Dioxide (kg/kg material)**	**Methane (kg/kg material)**	**Carbon Dioxide (kg/kg material)**
Dimensional Lumber and Engineered Wood	0.064	0.21	0.022	0.33
Cardboard	0.12	0.40	0.042	0.60
Gypsum Drywall	0.010	0.034	0.0038	0.052
LCD	0.036	0.12	0.012	0.183

2.6 Landfill Gas and Leachate Collection and Treatment

Because the majority of US CDD landfills do not have a GCCS, an LCI process dataset that models the environmental burdens associated with gas collection and management at a CDD landfill was not developed. However, a complete MSW disposal LCI process dataset should include the materials and energy used and the emissions resulting from the construction, operation, and eventual decommissioning of the average nationwide GCCS installed at MSW landfills, since MSW landfills also accept CDD materials. An MSW landfill GCCS typically includes a flare station, which houses gas movers (i.e., blowers), destruction/treatment (e.g., flares, reciprocating internal combustion engines), and monitoring equipment. At the present time, data on the average amount of electricity necessary to extract and treat the LFG appear to be unavailable. Furthermore, no estimates of the specific composition of flare station materials and the energy required to install/decommission the components of a GCCS were found. The Project Team was unable to develop a generic process that models the average nationwide energy consumption per unit volume of LFG handled by GCCS equipment due to lack of data.

As of 2012, 17 states required that CDD landfills have liners and leachate collection systems (IWCS 2012). It was assumed that the typical US CDD landfill is unlined. However, the emissions resulting from installing, operating, and decommissioning a leachate collection and removal system (LCRS) at an MSW landfill need to be considered for CDD materials placed in MSW landfills. Two subcategories of emissions result from leachate collection and treatment: those based on the total volume of leachate handled (e.g., emissions resulting from electricity used to pump leachate) and those based on the treatment of specific leachate constituents (e.g., amount of energy necessary for aeration of leachate to reduce the concentration of BOD below regulatory limits).

While Ecobalance (1999) provides an estimate of the average leachate collected per ton of MSW deposited in MSW landfills at different intervals after placement (i.e., 20, 100, and 500 years), it does not estimate the average amount of electricity necessary to collect and transport leachate; the only energy consumption estimate provided is for the WWTP treatment of BOD, given as 0.001 kWh/g BOD removed. While WWTP removal efficiencies are provided for seven leachate constituents/parameter categories (i.e., COD, BOD, ammonia, phosphate, total suspended solids, heavy metals, and trace organics), the specific energy required for the removal of each of these is not provided. While the emission path for each parameter is specified (e.g., BOD treatment will release carbon dioxide to air, emit biomass sludge), an additional complicating factor is the potential for the sludge from the WWTP to be disposed of at the MSW landfill from which the leachate came. Furthermore, it does not appear that the removal efficiencies are temporally weighted to account for the average LCRS operational period (estimated as 40 years); to estimate the mass of a particular contaminant (or treatment byproducts of a particular contaminant) released to surface water or to air as a result of WWTP treatment, it would be necessary to estimate the mass emissions of the contaminant through MSW leachate over time. Because of these limitations and the data gaps identified in the next section, it was not possible to create an LCI dataset for the collection and treatment of leachate produced from disposing of CDD materials in an MSW landfill.

2.6.1 Data Gap Analysis of Landfill Gas and Landfill Leachate Collection and Treatment

To develop an LCI dataset that simulates emissions resulting from the use of an MSW landfill GCCS to collect gas produced from CDD materials, the following data gaps need to be addressed:

1. **Average energy required to construct and manufacture the components of a GCCS**. Based on the experience of the Project Team, significant quantities of steel are used in the manufacturing the flare stations used for a MSW landfill GCCS; the quantity of steel and other components used in manufacturing flare stations is important for developing a representative LCI for GCCS construction. Furthermore, determining the average energy associated with gas well drilling/trenching and GCCS pipe welding would allow for a more accurate estimate of the emissions associated with GCCS construction and installation.

2. **Average energy required to collect a unit volume of LFG**. Electricity is necessary to power GCCS equipment; therefore, emissions associated with powering GCCS equipment should be allocated to organic CDD materials on a gas-collected-per-mass-disposed-of basis.

3. **Data on common practices for decommissioning GCCS equipment**. Information on the EOL management of GCCS equipment is necessary to estimate the complete environmental burdens associated with its serviceable life. Depending on whether the flare station or other GCCS components are recovered or simply disposed of will have an impact on the total emissions resulting from the MSW landfill disposal process dataset.

Besides the leachate data limitations presented for each of the materials in its respective chapter, the following additional information would be necessary to develop a representative dataset for leachate emissions resulting from the placement of CDD in an MSW landfill:

1. **Volume of leachate collected over the serviceable life of an MSW landfill LCRS**. This volume would be necessary to estimate the average emissions associated with collecting and transporting MSW leachate, irrespective of leachate quality.
2. **Mass fraction of different C&D-material-produced leachate constituents, which may be expected to be released in leachate during the service life of the LCRS.** Currently, it is not possible to estimate the total mass fraction of different leachate parameters that will be collected and treated at a WWTP. Therefore, it is not possible to assign emissions to the correct environmental bins (e.g., air, surface water, groundwater). For example, it is expected that calcium would leach from the placement of demolished concrete in an MSW landfill environment. There currently appears to be no information that specifically predicts the total amount of calcium that will leach out from concrete placed in an MSW landfill over the operational life of an LCRS. Therefore, it is not possible to estimate the quantity of calcium removed or energy consumption associated with WWTP calcium removal. Similar to LFG production, it is likely that individual leachate contaminants will be released at different rates depending on the specific CDD material. Long-term leaching data specific to each CDD material will likely be necessary before an LCI

dataset representing the emissions resulting from collecting and treating leachate from MSW-landfill-placed CDD can be created.

3. **Energy necessary to collect, transport and treat leachate on a per-volume basis.** As mentioned previously, it is anticipated that a certain baseline electricity demand will be necessary for leachate collection, transport, and some treatment processes, independent of leachate quality.

4. **Energy and materials necessary to treat specific leachate constituents.** Different sub-processes of WWTPs are designed to remove different categories of contaminants (e.g., aeration for BOD, clarification for suspended solids, activated carbon filtration for metals and organic compounds). The energy and (as applicable) operation and maintenance consumables for each treatment sub-process need to be quantified on a per-mass-contaminant basis before a suitable MSW leachate treatment dataset can be developed to simulate the emissions associated with the treatment of individual contaminants. This estimate is necessary to be able to estimate the specific emissions resulting from treating leachate resulting from the placement of specific CDD materials in an MSW landfill environment.

5. **End location of contaminants removed from MSW leachate at a WWTP.** Ecobalance (1999) discusses three process emission categories from WWTPs: air emissions, surface water emissions, and solid (i.e. sludge) emissions. Once leachate is treated, it is necessary to know what fraction of the contaminants (or contaminant treatment byproducts) was emitted into which emission category. While removal efficiencies refer to the fraction of the contaminant removed from the leachate, data pertaining to partitioning of contaminant into sludge and air are not available. This is particularly important for those contaminants that may be removed through multiple treatment processes (e.g. BOD removal through suspended solid clarification versus aeration to carbon dioxide).

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3 Asphalt Pavement

3.1 Introduction

Asphalt pavement is constructed in multiple layers: top surface, intermediate, and base. The top two layers typically consist of approximately 95% aggregate and 5% asphalt (FHWA 2011). Asphalt (also referred to as *bitumen*), which is a product of petroleum refining and includes the denser fraction of crude oil hydrocarbons, is used as an aggregate binder. Aggregates used for asphalt pavement production may include gravel, sand, and crushed stone; crushed stone may include various rock types such as limestone, dolomite, and granite. Aggregate and bitumen are commonly combined at HMA plants at elevated temperatures prior to transport to a job site for pavement surface application. While there are numerous processes for paving mix production (e.g. hot mix, warm mix, cold mix), HMA production is the most commonly used process; approximately 94% of US roads are paved with HMA (US EPA 2012, Kelly 2011). NAPA (2013a) estimated that approximately 325 MMT of asphalt pavement were produced and used for road construction, rehabilitation, restoration, and resurfacing in the US in 2012.

Asphalt pavements are routinely rehabilitated, resurfaced, and reconstructed due to surface wearing over time. Asphalt pavement is removed by either road milling or demolition through excavation. Milling involves grinding the road surface using a machine which has a toothed rotary drum that can be lowered or raised to adjust the milling depth. Road pulverization and excavation may be used in instances where the road base is compromised and no longer provides sufficient structural support for the overlying layers or for instances where milling is not feasible or economically justified (e.g., parking lots, small road stretches). Pavement removed via milling is typically referred to as RAP; all pavement removed after service life is herein referred to as RAP.

Once removed, RAP may be recycled or disposed of. Recycling most commonly includes introduction into new HMA or use as an aggregate in a fill application (e.g., road base, embankment). Figure 3-1 identifies the flow of materials and processes that should be considered for conducting an LCA of asphalt pavement EOL management. Upstream processes such as primary material extraction, crude oil refining, aggregate production, and HMA production are also needed for an EOL LCA, as a majority of RAP is recycled in a closed loop to produce paving mix.

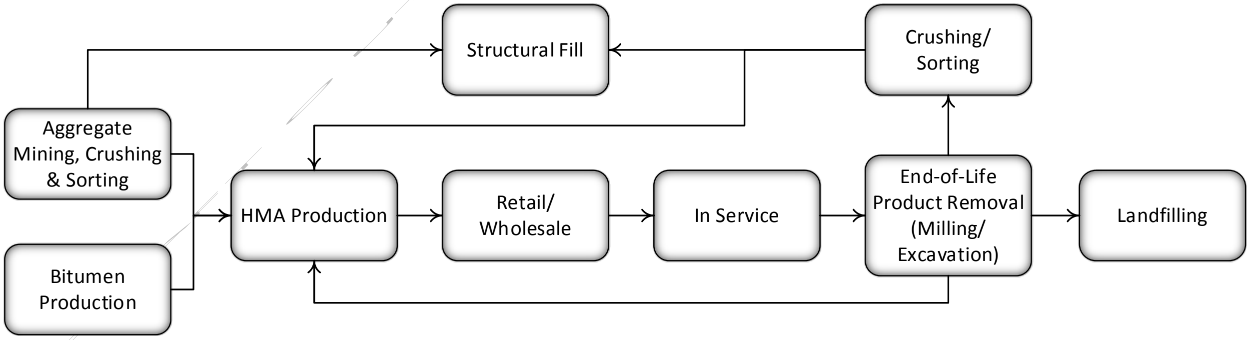

Figure 3-1. Materials Flow for HMA Production and EOL Phase Management

3.2 Management at EOL

A significant amount of RAP generated in the US results from pavement resurfacing, rehabilitation, and reconstruction operations (Copeland 2011). Based on a nationwide survey in 2011 of the asphalt pavement industry, NAPA estimated that approximately 65.7 MMT of RAP were used by the asphalt pavement producers (NAPA 2013b). In an independent survey for the same year, the USGS estimated that approximately 13.4 MMT of RAP were recycled by construction aggregate mining companies (e.g., construction crushed stone and sand and gravel producers) and construction and demolition companies (Bolen 2013). US EPA (2014a) estimated that approximately 4.3 MMT of RAP was disposed at permitted or registered solid waste management facilities. Together, it appears that nearly 83.5 MMT of RAP was generated in 2011.

Figure 3-2 presents the distribution of RAP used in different applications in the US in 2011 based on the US EPA (2014a) estimate of RAP landfilled by permitted or registered solid waste disposal facilities and surveys by NAPA (2013b) and Bolen (2013). As Figure 3-2 shows, more than 70% of RAP generated in the US in 2011 was used in HMA/WMA production. Approximately 12% of the pavement produced in 2010 was warm mix asphalt (WMA) (FHWA 2012), so it is anticipated that the bulk of RAP was used in HMA applications. As presented in Figure 3-2, "Other uses" of RAP include cold mix production and untracked use. Aggregate uses may include application as a road base course, fill material for road embankments, and other fill applications.

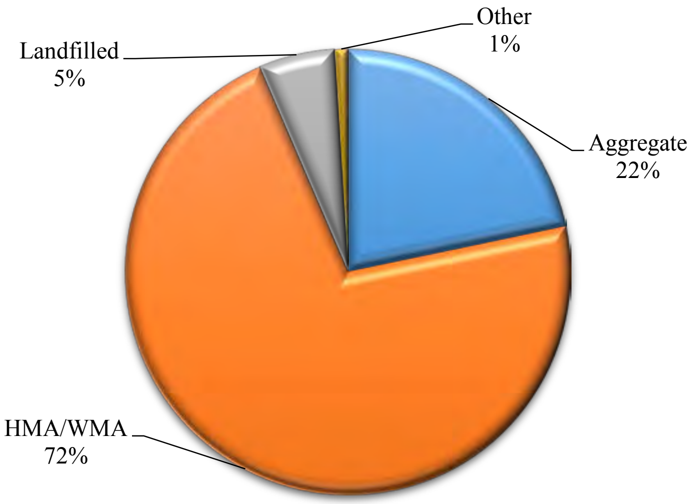

Figure 3-2. Distribution of RAP Uses in 2011

Table 3-1Table 3-1 lists the processes that should be considered to conduct an LCA of EOL management of asphalt pavement. The emissions associated with energy and materials requirements and process non-energy emissions (e.g., fugitive dust, liquid emissions associated with disposal of RAP in a landfill) were taken into account for compiling the different LCI datasets.

Table 3-1. Asphalt Pavement EOL Management Process Descriptions

Process	Description
Crude Oil Extraction	Extracted crude oil is separated from water and transported to a refinery typically by means of tankers or pipelines.
Asphalt Production and Storage	Asphalt is produced as a co-product of petroleum refining. Following production, asphalt is typically stored at elevated temperatures for it to remain a liquid.
Aggregate Production	Aggregate production generally includes the mining and processing of stone, gravel, and sand. These materials are typically ground and

Process	**Description**
	fractionated at the mining site before shipment to the end-user. Additional information on aggregate production is provided in Chapter 2 of the report.
HMA Production	Aggregates, asphalt, and sometimes RAP are mixed at elevated temperatures to produce HMA. Because asphalt solidifies at ambient temperatures, it is typically stored hot at the plant until added with aggregates to produce HMA.
Transport	While transport LCI are generally presented in units of mass-distance (kg-km), regional average distances between processes is necessary for the development of a regional-level LCA.
Asphalt Pavement Removal	Pavements are removed either through milling or excavation at the end of their serviceable life. A milling machine has a rotating drum with studded teeth that grind down the surface of the pavement to a pre-determined depth. The ground-up material is discharged as millings. Pavement excavation involves the use of heavy equipment to break up the pavement into chunks prior to removal.
Landfill Disposal	Following removal, asphalt pavement may be disposed of at a CDD, inert, or sanitary (MSW) landfill.
RAP Processing	RAP processing may include additional crushing and fractionation (i.e., sorting into different size categories). The extent of RAP processing is dependent on many factors such as end use, amount of RAP used relative to primary materials for HMA, production method, and the duration of RAP storage in a stockpile.
RAP Use as Aggregate	RAP may be used as a primary aggregate substitute in a variety of fill applications.

3.3 LCI Sources

Peer-reviewed literature, government and private industry publications, and various LCA modeling tools were reviewed to identify available LCI datasets pertaining to asphalt pavement EOL management processes. Table 3-2 lists data sources reviewed to develop the LCI presented in this chapter. If LCI data were not available, process metadata and documentation (e.g., included emission categories, background data used to compile the dataset, geographic location and time period of the data) were reviewed to evaluate the completeness of the dataset. If available, the primary sources of information used to develop the LCI datasets and information were reviewed.

Table 3-2. List of Sources Reviewed for LCI Data

LCI Source	**Description**
Athena (2001)	Franklin and Associates developed LCI for Road and Roofing Asphalt, which include emissions associated with crude oil extraction and processing at a petroleum refinery for asphalt production (i.e., cradle to gate). The LCI data were compiled for the report presented to Athena in 1999 and are representative of asphalt production in the US.
US LCI (2012)	The National Renewable Energy Laboratory has published an LCI database which includes datasets for a wide variety of services and material, component, and assembly production processes within the US.

LCI Source	Description
Wilburn and Goonan (1998)	The authors provide energy requirements associated with crushing/sorting stone, sand and gravel, and RAP. These data were taken from the Portland Cement Association and an energy audit of a recycling facility in Denver, Colorado.
NRC (2005)	Natural Resources Canada developed a Road Rehabilitation Energy Reduction Guide for Canadian Road Builders based on a survey of pavement producers. This report provides fuel-use data at asphalt pavement plants in the Canadian context.
Stripple (2001)	Stripple (2001) presents LCI for a wide variety of road construction, pavement production, maintenance, and demolition activities (e.g., land clearing activities for road placement, installation of signs). Emissions are based on information gathered from 1990-1994 for numerous road manufacturing and upkeep processes taken from a variety of industry and heavy equipment manufacturer sources.
Eurobitume (2012)	The European Bitumen Association provides LCI of the bitumen (i.e., asphalt) production process from crude extraction to hot storage of bitumen at the refinery site.
WARM	EPA’s Waste Reduction Model presents data on GHG emissions associated with source reduction, transport, recycling, and landfilling (i.e., collection and placement) of asphalt pavement.
AP-42 (US EPA 1995a)	Provides air emissions data for blasting agent detonation (for aggregate mining), HMA plants, sand and gravel and crushed stone processing (PM emissions only), and petroleum refinery processes
NIST BEES	The National Institute of Standards and Technology Building for Environmental and Economic Sustainability model allows for an economic and environmental impact comparison among various building materials.
ASMI IE	The Athena Sustainable Materials Institute developed the Impact Estimator for Highways, which is LCA software for the evaluation of the environmental implications of different roadway designs.
GaBi	GaBi includes an LCI database developed by PE international that contains US-specific datasets related to crude oil, aggregate (i.e., limestone), and asphalt production.
Ecoinvent	Ecoinvent is an LCI database developed by the Swiss Centre for Life Cycle Inventories, which includes specific processes related to the EOL management of numerous individual materials. It includes processes related to the disposal of waste asphalt pavement.

3.4 LCI Related to HMA Pavement Production

3.4.1 Raw Materials Extraction

3.4.1.1 Aggregate Mining, Crushing and Sorting

Crushed rock and gravel and sand are the most commonly used aggregates in asphalt pavement production, although a wide variety of industrial byproducts (e.g., glass, crumb rubber, ash, steel slag) are also occasionally used (NAPA and EAPA 2011). According to Willett (2013), approximately 80% of crushed stone consumption (which was tracked by use) was used “mostly for highway and road construction and maintenance.” Also, approximately 13% of the 810 MMT of sand and gravel produced in the US in 2011 which was categorized by end use was used for asphalt pavement aggregates (Bolen 2013). Additional information on the development of LCI datasets for the production of different aggregates is included in Chapter 2 of the report.

3.4.1.2 Asphalt Production

Asphalt is a co-product of the petroleum refining process and is used as the binding agent to provide aggregate cohesion. Approximately 85% of the asphalt produced worldwide is used for asphalt pavement production (NAPA and EAPA 2011). Approximately 21 MMT of asphalt and road oil were produced in the US in 2012 (US EIA 2014). The LCI for the asphalt production process include emissions associated with crude oil extraction, transportation, and refinery processing.

The US LCI (2012) database already includes a “crude oil, at production” process which cites 2003 information developed by Franklin Associates. However, this source document was not found. Franklin Associates also developed the crude oil LCI dataset used in Athena (2001), which uses information from 11 sources dated between 1962 and 1996. The GaBi database references 76 individual sources for its US-specific “Crude Oil Mix” process dataset. While LCI datasets with information on crude oil production were available in other publications and LCA models, these were representative of international crude mixes.

Crude oil refineries use a number of physiochemical processes to refine crude into a variety of petroleum products; however, the production of asphalt usually consists of desalting, atmospheric and vacuum distillation, and deasphalting (Athena 2001). Desalting involves water washing the crude to remove any water-soluble constituents. Atmospheric distillation classifies crude into those constituents that have a boiling point higher than 650 degrees Fahrenheit. These constituents pass to the vacuum distillation process. Vacuum distillation allows a lower temperature classification of the residuals from atmospheric distillation. Deasphalting occurs when the residuals from vacuum distillation are removed through the use of a liquid hydrocarbon solvent. The steam that is stripped from this solvent is asphalt (Athena 2001).

Athena (2001) and US LCI (2012) provide LCI information for asphalt production in the US; information in Athena (2001) was used to assess the greenhouse gas emissions and energy requirements corresponding to asphalt production as modeled by WARM. GaBi also contains a US-based asphalt production process titled “Bitumen at refinery” which also includes inventory data resulting from crude oil extraction. Eurobitume (2012) and Stripple (2001) provided asphalt production LCI for Europe and Sweden, respectively. The US LCI (2012) LCI database includes a “Petroleum refining, at refinery” process which produces asphalt and uses inventory emissions information from LCI datasets published by Franklin Associates in 2010.

Eurobitume (2012) provides a detailed LCI of numerous sub-processes within the asphalt production industry of Europe, citing source materials obtained from industry contacts, a Eurobitume survey, and other sources from 2008-2010. LCI data include the emissions and energy consumption from crude oil extraction to the hot storage of asphalt onsite. It is important to note that Eurobitume (2012) includes emissions from the construction/manufacturing of asphalt production infrastructure.

Asphalt is generally stored hot at the refinery until it is either transported to an asphalt terminal (often by rail) or by tanker truck (depending on distance) to an HMA plant for use. Transportation distances and timeframes are critical in order to keep the asphalt at a high enough temperature to ensure a sufficiently low viscosity so that the material can be pumped from the transport vehicle into heated storage tanks at the HMA plant (Walker and Davis 2008, Astec 2009). In addition to hot storage at HMA plants, some major petroleum companies and asphalt pavement producers store hot asphalt at asphalt terminals (e.g., BP, Shell, Marathon, Associated Asphalt, C.W. Matthews). Terminals have large heated storage tanks that are used as a centralized location to hold asphalt until delivery to individual HMA plants. Some asphalt terminals use steam to heat rail car asphalt tanks so that the solidified material can be withdrawn and stored at the terminal (Walker and Davis 2008, Astec 2009). However, it does not appear that the US asphalt production LCI discussed above includes the process energy and process non-energy emissions from the hot onsite storage of asphalt at terminals. Depending on typical asphalt storage practices and durations, the absence of this information may have major limitation with the use of the "Petroleum refining, at refinery" dataset for quantification of environmental impacts of this process.

The US EPA (2006) provided equations for determining emission factors for the storage of organic liquids based on numerous parameters such as liquid properties, environmental conditions, tank type, and tank dimensions. However, it would be difficult to develop an average nationwide value for each of these variables to create a representative LCI dataset including emissions from the asphalt terminal storage, especially considering the absence of information with respect to the quantity of asphalt stored at terminals and the duration of storage. Also, the US EPA (2006) does not provide procedures for estimating emissions resulting from fuel consumption to keep asphalt storage tanks at elevated temperatures. Since the US LCI (2012) dataset includes publicly-available LCI for crude oil extraction and asphalt production, these LCI for asphalt pavement EOL management LCA are proposed for use in this report.

3.4.2 Transport

A wide variety of transportation modes, including ocean, road, and rail, are used to move numerous materials, including crude oil, asphalt, aggregate, pavement mix, and RAP. The emissions associated with crude oil transport are accounted for in US LCI (2012) petroleum refining processes. The LCI data for fuel-related emissions from various pieces of transport equipment are included in the US LCI (2012) database. However, the emissions associated with the manufacturing/building of transport equipment do appear to be included in these LCI. The transport distances associated with the different stages of asphalt pavement production, which would be region specific, are needed to estimate the emissions associated with the transport of other materials (besides crude oil). Wilburn and Goonan (1998) provide transport energy requirements for aggregate, sand and gravel, and reclaimed asphalt pavement in terms of joules per kg-km.

The Athena Sustainable Materials Institute (ASMI) Impact Estimator for Highways model allows the user to specify distances between various cradle-to-gate processes in the complete pavement production and placement process, or the user may choose default values. The model uses a database with numerous pieces of road-manufacturing equipment, each of which includes default operating specifications such as primary and secondary fuel consumption rates, production rates, load factors, and daily operational time. However, model defaults are specific to Canada – the opening input page for a new project only allows selection of a Canadian province for the project location. The model and accompanying documentation do not provide the sources of transport LCI due to LCI's proprietary nature.

In its environmental impact assessment of product use, the National Institute of Standards and Technology (NIST) Building for Environmental and Economic Sustainability (BEES) model considers the transport of products between the product manufacturer and the end-user. The model allows the user to input information for the distance between the product manufacturer and the use of the material, although the model already has a default value included for this distance. However, NIST (2011) does not provide details on the source of these default distances already included in BEES.

The US EPA (2012) uses crude oil transport information from the US LCI (2012) database and NRC (2005) for transport-related emissions of HMA pavement materials (e.g., asphalt and aggregates to the HMA plant, HMA to the road site) for the WARM model. NRC (2005) provides average distances (in Canada) from aggregate and asphalt production locations to HMA plants and the distance from HMA plants to road sites.

Distances for asphalt transport were calculated in the same way they were estimated for primary aggregates in Chapter 2 of this report, using transport data provided in USCB (2010) for the “Coal and Petroleum Products” commodity. Table 3-3 summarizes the US-wide average transportation distances of asphalt, HMA, and RAP materials.

Table 3-3. Transport Distances of Different Pavement Materials

Material Transport	Source and Representative Commodity	Total Amount Transported by Single Mode (million tons)	Mode	Ton-miles (in millions)	Average Transport Distance (miles)	Average Transport Distance (km)
Asphalt Transport from Refinery to Paving Mix Plant	USCB (2010)- Coal and Petroleum Products	459	Truck	33,900	73.9	119
			Rail	52,600	115	185
			Water	18,600	40.6	65.3
Paving Mix Transport from Paving Mix Plant to Road Sites						20

3.4.3 HMA Plants

HMA plants accept asphalt and specific gradations of aggregate (e.g. sand, gravel), heat and mix the ingredients using a drum or batch process, and store the mix at a sufficiently elevated temperature until it can be loaded for transport and laid as pavement at the roadway job site. Different gradations of aggregate are fed into HMA plant rotary dryers that operate at approximately 300 °F. Aggregates are dried and mixed with asphalt in the same rotary drum used for drying in continuously mixed plants, whereas aggregates dried in a rotary drum are discharged, screened, weighed, and then mixed with asphalt in a pug mill in a batch mix process (NAPA and EAPA 2011).

The environmental emissions from HMA production include particulate emissions from on-site aggregate handling, aggregate drying, RAP crushing/screening, process energy emissions (e.g., aggregate drying, asphalt storage tank heating, HMA storage heating), air emissions from the storage of hot asphalt and pavement mix (e.g., VOCs), and liquid emissions (e.g., stormwater run-off).

Stripple (2001) provides LCI data for HMA plant operation in Sweden, but it appears that only emissions associated with electricity and energy consumption are included in the dataset. Fuel-use data are reported to be based on operational information from a single HMA plant, and additional information is not provided. HMA energy requirements are also provided by NRC (2005), which surveyed five Canadian road builders to gather information on the energy requirements for pavement manufacturing and road rehabilitation. The US EPA (2012) used data reported by NRC (2005) to develop energy and emission factors for HMA plants in WARM.

The Manufacturing Energy Consumption Survey conducted by the US Energy Information Administration provides US-wide energy used by the asphalt paving mixture and block manufacturing industry (NAICS code 324121) by fuel type. The survey is conducted every 4 years and the most recent survey presents energy consumption data for 2010 (US EIA 2013a).

AP-42 (US EPA 2004) provides air emissions (gaseous and particulate matter) from HMA plants specific to plant type (i.e., drum versus batch mix) and in some cases fuel use (e.g., No. 2 fuel oil, No. 6 fuel oil, natural gas). The majority of emissions are provided per ton of HMA produced. However, emissions as a result of truck load-out, silo filling, and asphalt storage are based on equations that use the input of site-specific factors, such as asphalt volatility and HMA temperature, though default values for these parameters are provided in the document.

Asphalt paving industry fuel consumption data, asphalt pavement production data, HMA plant particulate emissions data, and aggregate and asphalt transport data were analyzed to develop an HMA production process LCI dataset as presented in Table 3-4. Emissions are provided per kilogram "Asphalt pavement, at production" flow. The industry-wide fuel-specific energy consumption data for the asphalt paving mixture and block industry (NAICS 324121) compiled for 2010 by the US EIA (2013a) along with the total pavement production data for 2010 compiled by NAPA (2013a) were used to estimate the energy requirement for producing one kilogram of pavement. The energy associated with the "Other" fuel category was assigned to natural gas because on an equal-energy (btu) basis, natural gas constituted 72% of the total fuel consumed by the asphalt paving mixture and block manufacturing industry.

Particulate emissions data were taken from AP-42, Hot Mix Asphalt Plants (US EPA 2004) since HMA production constitutes the majority (~80%) of national asphalt pavement production (NAPA 2013b). Particulate emissions were modeled as the average uncontrolled emissions from a drum mix plant dryer and a batch mix plant dryer, hot screens, and mixer. AP-42 only provides particulate matter emission factors for the dryer, hot screening, and mixing process and does not account for particulate matter emissions from aggregate stockpiles, aggregate loading, aggregate conveyance, and onsite equipment movement. Due to this limitation, uncontrolled particulate emission factors reported by AP-42 were selected for modeling particulate emissions from the HMA plant. While particulate emissions do result from dryer fuel combustion, it is expected that these only make a minor contribution to the overall particulate emissions released from drying and mixing, particularly since natural gas is the primary fuel used by HMA plants, as described above.

Polycyclic aromatic hydrocarbons (PAHs) have been reported to be one of the major classes of air pollutants emitted from HMA facilities (US EPA 2000). Based on a US EPA estimate, approximately 13 lb of PAHs are emitted annually from a typical batch mix HMA facility with an annual production rate of 100,000 tons of HMA (US EPA 2000). Lee et al. (2004) conducted a study to quantify PAHs emissions from batch HMA plants. Gas samples were taken from batch mixers, preheating boilers, and discharging chutes. The reported PAH emission factor for batch mix plants was 139 mg/ton (30.6 lb per 100,000 tons) of product (Lee et al. 2004). Lee et al. (2004) also reported that approximately 90% of carcinogenic PAHs were removed by air-pollution-control equipment at the HMA plants studied.

AP-42 does provide equations for estimating non-fuel related VOCs emissions from HMA silo filling and load-out based on site-specific variables such as temperature and asphalt volatility data. However, the individual VOC emissions could not be estimated using the procedure outlined in AP-42 without making a variety of assumptions. The VOC emissions were, therefore, not estimated and included in Table 3-4. The absence of VOC emissions information in the dataset may be a major limitation in assessing the environmental impacts associated with the asphalt pavement production process. It should also be noted that due to lack of data, stormwater emissions associated with aggregate and RAP stockpiles and emissions from plant construction and component manufacture were not included in Table 3-4.

Table 3-4. Proposed LCI Dataset: Asphalt Pavement Production, Average Energy Mix

Input Flow	Source	Category	Unit	Amount
Limestone, at mine	US EPA (2012)		kg	0.95
Bitumen, at refinery	US EPA (2012)		kg	0.05
Electricity, at industrial user	US EIA (2013a)	Flows	kWh	0.0033
Diesel, combusted in industrial boiler	US EIA (2013a)	Flows	L	0.001
Natural gas, combusted in industrial boiler	US EIA (2013a)	Flows	m3	0.0042
Transport, barge average fuel mix	USCB (2010)		t*km	0.0155
Truck transport, class 8, heavy heavy-duty (HHD), diesel, long-haul, load factor 0.75	USCB (2010)		t*km	0.0697
Transport, train, diesel powered	USCB (2010)		t*km	0.0277
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Asphalt pavement, at production		*Construction and Demolition Debris Management*	*kg*	*1*
Particulates, < 2.5 um	US EPA (2004)	air/unspecified	kg	0.000443
Particulates, > 10 um	US EPA (2004)	air/unspecified	kg	0.0123
Particulates, > 2.5 um, and < 10um	US EPA (2004)	air/unspecified	kg	0.00228

3.5 LCI Related to Disposal

Various literature sources suggest different rates of landfill disposal of asphalt pavement. Wilburn and Goonan (1998) estimated that about 20% of asphalt pavement is disposed of in landfills. A 2011 NAPA (2013b) survey of RAP use, which included 203 asphalt mix producing companies from 49 states representing 1091 plants, suggested that less than 1% of all RAP is disposed of in a landfill. A compilation of the data from the USGS, NAPA and permitted CDD materials processing and disposal facilities compiled by US EPA (2014a) from across the US suggests that approximately 5% of RAP is landfilled.

Air emissions from landfill disposal of asphalt pavement will result from the operation of landfill equipment during material and cover soil compaction and placement, including both fuel-related and pre-combustion emissions. RAP exposure to precipitation or other liquids (e.g., landfill leachate) is expected to result in leached emissions and the emissions are expected to depend on the biogeochemical environment (e.g., MSW landfill, CDD landfill). Only a single Ecoinvent LCI dataset (2014) appears to include leached emissions from waste pavement.

Leachable emissions from RAP were estimated using the SPLP (batch test) and leaching column data reported by Townsend and Brantley (1998). Townsend and Brantley (1998) conducted batch and column leaching tests on asphalt pavement samples collected from six sites. Batch test data were used for contaminants except heavy metals. Batch test concentrations were multiplied by the total solution volume and divided by the sample mass to estimate leachability on a per-kilogram-asphalt pavement basis. Contaminant emissions were not estimated for parameters below the detection limit in more than half (i.e., three) of the samples; bromide, sodium, and potassium were excluded. The detection limit was used as the concentration for measurements below detection for samples in which the contaminant emission was quantified. Nitrate and sulfate data reported by the study were not used since the SPLP extraction fluid contains these anions. The emissions of non-purgeable organic compound were not estimated as this compound is not included as an elementary flow in US LCI (2012). Total dissolved solid (TDS) data reported by the study were also not included to avoid double counting of emissions as some of the contaminants listed in Table 3-5 are included in the TDS measurement.

None of the measured heavy metals and organic compounds leached above the detection limits in batch leaching tests. Therefore, column test data were used to estimate heavy metals and organics emissions. All the organic compounds (VOCs and PAHs) and heavy metals except lead targeted in this study were below detection in the column tests. The total lead leached in saturated column test was greater than that of the unsaturated column test; the lead emission was estimated based on the saturated column test data, which corresponds to a L:S ratio of 0.257. Lead emission was estimated for each of the six samples by dividing the total amount of lead leached from RAP sample by the total RAP amount used for the test. The average lead leached from six samples was estimated to be 6.58 microgram per kg of RAP.

Azah (2011) conducted batch and (saturated and unsaturated SPLP) column leaching tests on RAP samples collected from five Florida locations to assess PAH leaching. Leaching data were selected from column tests for PAHs which were detected in over half of the samples. Four PAH's were detected in over half of the samples during unsaturated column testing (i.e., fluoranthene, pyrene, benzo(k)fluoranthene, benzo(a)pyrene) while three PAHs were detected in over half of the samples during saturated column testing (i.e., pyrene, benzo(g,h,i)perylene, benzo(b)fluoranthene). Because the final L:S was higher for saturated column testing compared to unsaturated column testing (i.e., approximately 2 versus 1.2, respectively) saturated column leaching results were selected over unsaturated column testing results for pyrene since this compound was detected in over half of the samples for both column testing conditions. Data were used to estimate liquids emissions from the disposal of RAP in CDD material landfills. Column leaching data were used to estimate the total leachable amount of PAHs – all below-detection-limit (BDL) measurements subsequent to detected concentrations were excluded from the analysis. The other BDL measurements were included at the detection limit concentration. Temporal column PAH concentrations (ng/L) were multiplied by the L:S ratio (L/kg) of the solution and values were summed for each PAH to estimate the PAH's total leachability on a per-kilogram-asphalt-pavement basis.

The energy use and the associated emissions from landfill operation (e.g., waste placement, compaction) include diesel use in heavy equipment and electricity use in landfill buildings (e.g., administrative buildings, workshop). In the absence of additional data, it was assumed that asphalt pavement would be transported 20 km for landfill disposal. Diesel consumption from landfill operations and electricity consumption from landfill administrative offices and workshop areas were estimated from Ecobalance (1999) and IWCS (2014b), respectively, and these flows are included as the "CDD landfill operations" input flow, as detailed in Chapter 2. Details on how cover soil was assigned for the placement of demolished asphalt pavement in a CDD landfill is also included in Chapter 2, and is based on the bulk density of asphalt pavement as provided in CCG (2006). RAP leaching data, energy consumption data from landfill operations, and the assumed transport distance were used to develop an LCI process dataset for the disposal of RAP at an unlined inert debris landfill, as presented in Table 3-5. Emissions are provided per kilogram "Asphalt pavement, at unlined CDD landfill" flow.

Table 3-5. Proposed LCI Dataset: Asphalt Pavement, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
Asphalt pavement, recovered from milling		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.0414
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Asphalt pavement, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Calcium, ion	Townsend and Brantley (1998)	Water/Groundwater	mg	273
Chloride	Townsend and Brantley (1998)	Water/Groundwater	mg	70.3
COD, Chemical Oxygen Demand	Townsend and Brantley (1998)	Water/Groundwater	mg	2230
Fluoride	Townsend and Brantley (1998)	Water/Groundwater	mg	23.9
Lead	Townsend and Brantley (1998)	Water/Groundwater	µg	6.58
Magnesium	Townsend and Brantley (1998)	Water/Groundwater	mg	26.7
Fluoranthene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	49.3
Pyrene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	34.6
Benzo(k)Fluoranthene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	2.55
Benzo(a)pyrene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	5.74
Benzo(g,h,i)perylene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	54.9
Benzo(b)fluoranthene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	3.07

3.6 LCI Related to Recycling

3.6.1 RAP Processing

RAP may need to be processed prior to recycling as an aggregate or for the production of new paving mix. The extent of RAP processing would be dependent on a number of factors, such as the pavement-removal method (i.e., milling, demolition and excavation), duration it is stockpiled, and end-use specifications. For those instances where RAP is produced as a result of road demolition and excavation, RAP must be size-reduced before it can be recycled into new HMA or for most aggregate applications. Additional processing is also likely to be required when RAP constitutes greater than 20% of the new paving mix for the mix to meet specified fractionation requirements. According to Brock and Richmond (2007), many continuously fed HMA plants will have a closed-circuit crushing system on the front end of the aggregate feeder where oversized milled material and aggregate is screened away from the feed, crushed, and then returned to the screen. Emissions from crushing and screening RAP millings could in this case be considered part of the total emissions released from the HMA plant. However, it is possible that RAP produced as a result of pavement demolition and excavation may be crushed offsite from an HMA plant.

The emissions from crushing/sorting equipment include those associated with materials (e.g., equipment, consumables), energy (fuel) inputs, and particulate matter released during processing. While particulate emissions information for asphalt pavement crushing/sorting was not located, Wilburn and Goonan (1998) reported energy requirement of 16.5 MJ for crushing/sorting 1 MT of recycled asphalt pavement based on data from the Portland Cement Association and data from a recycling facility in Colorado. In the absence of additional data, electricity and diesel consumption were each assumed to constitute 50% of the total energy requirements. A similar approach was used by the US EPA (2003) for estimating emission factors for demolished PCC processing for WARM. Energy units were converted into electricity and diesel quantities assuming 3,412 btu/kWh and 138,690 btu/gallon diesel, as provided by US EIA (2013b).

Table 3-6 presents the proposed LCI dataset for RAP processing. Similar to aggregate crushing and sorting during production, it is expected that particulates would be a source of emissions released from RAP crushing and sorting operations. However, due to a lack of data, fugitive dust emissions released during the grinding process and emissions from manufacturing, maintaining, and disposing of/dismantling the grinding equipment are not included in Table 3-6. If recovered pavement is not processed prior to use, the diesel and electricity consumption should be excluded from the LCA. In the absence of national average data on the distance from the road demolition site to RAP processing, a distance of 20 km was assumed with transport by a single-unit, short-haul, diesel-powered truck.

Table 3-6 Proposed LCI Dataset: Reclaimed Asphalt Pavement, at Processing Plant

Input Flow	Source	Category	Unit	Amount
Asphalt pavement, from road demolition		Construction and Demolition Debris Management	kg	1
Diesel, combusted in industrial equipment	Wilburn and Goonan (1998)	Flows	L	0.000213
electricity, at industrial user	Wilburn and Goonan (1998)	Flows	kWh	0.00229
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.02
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Reclaimed asphalt pavement, at processing		*Construction and Demolition Debris Management*	*kg*	*1*

3.6.2 RAP Use in HMA

Because RAP contains valuable asphalt binder and aggregate, its use in new HMA to replace primary aggregates and binders has been perceived as the most economical use of RAP (Copeland 2011); this use also avoids the environmental impacts from the primary production and transport of both materials. Milled

RAP, about 75% of which has a typical diameter less than 0.5 in., may often be added without any further processing besides screening and crushing of oversized material (Brock and Richmond 2007). However, mixes using more than 20% RAP need additional RAP crushing and sorting to fractionate the material according to the mix design.

Aurangzeb et al. (2014) and COLAS (2003) suggest that the energy necessary for HMA production is not related to RAP content for mixtures using less than or equal to 50% RAP. However, information on whether RAP content has an effect on HMA plant non-fuel emissions was not discovered. Additional information from a couple of French studies suggests that RAP content may impact air (e.g., volatile organic compounds, polycyclic aromatic hydrocarbons, odor) and liquid (e.g., total hydrocarbons and polycyclic aromatic hydrocarbons) emissions from pavement (Jullien et al. 2006, Legret et al. 2005).

3.6.3 RAP Use as Aggregate

RAP produced from road demolition and excavation activities will likely need to be crushed prior to use in structural fill applications; however, it is possible that millings may be used without additional processing. The primary emissions resulting from the use of RAP as a fill material include leaching to groundwater (Townsend and Brantley 1998, Azah 2011). The use of RAP as aggregate would avoid the emissions resulting from the production and transport of primary aggregates as presented in Chapter 2. Table 3-7 presents LCI for RAP transport and use as aggregate. It was assumed that the processed RAP will be transported 20 km from the processing site to the end-use site.

Table 3-7. Proposed LCI Dataset: Reclaimed Asphalt Pavement, Use as Fill

Input Flow	**Source**	**Category**	**Unit**	**Amount**
Reclaimed asphalt pavement, at processing		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Reclaimed asphalt pavement, use as fill		*Construction and Demolition Debris Management*	*kg*	*1*
Calcium, ion	Townsend and Brantley (1998)	water/groundwater	mg	273
Chloride	Townsend and Brantley (1998)	water/groundwater	mg	70.3
COD, Chemical Oxygen Demand	Townsend and Brantley (1998)	water/groundwater	mg	2230
Fluoride	Townsend and Brantley (1998)	water/groundwater	mg	23.9
Lead	Townsend and Brantley (1998)	water/groundwater	µg	6.58
Magnesium	Townsend and Brantley (1998)	water/groundwater	mg	26.7
Fluoranthene	Azah (2011)	Construction and Demolition Debris Management/ groundwater	ng	49.3
Pyrene	Azah (2011)	Construction and Demolition Debris Management/ groundwater	ng	34.6

Benzo(k)Fluoranthene	Azah (2011)	Construction and Demolition Debris Management/ groundwater	ng	2.55
Benzo(a)pyrene	Azah (2011)	Construction and Demolition Debris Management/ groundwater	ng	5.74
Benzo(g,h,i)perylene	Azah (2011)	Construction and Demolition Debris Management/ groundwater	ng	54.9
Benzo(b)fluoranthene	Azah (2011)	Construction and Demolition Debris Management/ Groundwater	ng	3.07

3.7 Data Gap Analysis and Opportunities for Additional LCI Data

The Project Team reviewed LCA software models, government and industry publications, and peer-reviewed literature to determine the availability of US-based asphalt pavement management LCI data. Three models and three publications were identified that contained at least partial emissions data for some portion of asphalt management. With the exception of WARM and NREL (WARM uses information from a limestone mining process included in the 2009 NREL database), each of these data sources is independent of each other, (i.e., the same primary data are not being used in multiple sources). As shown in Table 3-8, WARM process data are partial because WARM only analyzes GHG emissions. Similar to WARM, AP-42 only includes partial data because the datasets only include process air emissions. Wilburn and Goonan (1998) only provide energy requirements for RAP processing, without distinguishing what fuels are used to provide this energy. Townsend and Brantley (1998) used SPLP batch and column testing to estimate the leachability of numerous organic, inorganic, and metal parameters. However, the leaching data are only considered partial because testing occurred in 1997-1998, and it appears that some laboratory detection limits have significantly decreased since that time; for example, Azah (2011) reports the leachable amount of PAHs from RAP, where the detection limit for these tests ranged between 0.0001 and 3.5 µg/L while the detection limit for Townsend and Brantley (1998) ranged between 0.5 and 5 µg/L. Also, Townsend and Brantley (1998) column testing occurred over 40 days while column testing by Azah (2011) occurred over 35 days. Additional extended runs would be necessary to observe concentration trends to estimate the total leachable concentrations from RAP placed in a fill or unlined landfill. Data from Azah (2011) are partial because Azah only presents the leachable amounts of PAHs. All identified models and publications that include environmental burdens with respect to some portion of asphalt pavement management are presented in Table 3-8. The table shows that LCI data are better documented for primary material extraction and HMA production processes than for pavement removal and EOL management.

Based on a review of these sources, the following needs for additional US-specific LCI data with respect to the EOL management of asphalt pavement were identified:

1. **A complete inventory of environmental burdens associated with constructing, operating, and decommissioning an HMA plant**. AP-42 estimates emissions for a number of sub-processes at HMA plants, where emission factors are categorized by plant type and fuel type (US EPA 2004). Specific air emission datasets could be developed for batch and drum mix plants, categorized by whether they operate on natural gas, No. 2 fuel oil, No. 6 fuel oil, or waste oil. However, information on total HMA production from each of these plant types would be necessary to develop individual datasets representative of nationwide average emissions. Furthermore, information on particulate emissions from HMA plants is limited to actual measurements of a very small fraction of HMA plant processes. It appears that particulate emissions from onsite aggregate loading, discharging, conveyance, and stockpiles have not been assessed. Also, LCI data associated with HMA plant construction and component manufacture, as well as data related to plant

decommissioning were not found in literature. Besides air emissions provided by AP-42, air emissions are compiled by the US EPA Office of Air Quality Planning and Standards every three years for a variety of industries and are organized by NAICS code (US EPA 2014b). The most recent (2011) point source emissions inventory has nearly 1.9 million air emission measurements. It would be possible to aggregate the total emissions of each parameter for all point sources within an industry across the nation and correlate these emissions with national asphalt production data to develop an additional alternative average US HMA plant air emissions inventory for each ton of HMA produced. However, a portion of these emissions is associated with energy use, which is accounted for using such EPA LCI processes as "diesel combustion in industrial equipment" and "natural gas combustion in industrial boiler" included in the proposed LCI for HMA.

2. **Milling/Excavation equipment fuel consumption, operation, and manufacturing emissions.** The US EPA NONROAD model simulates air emissions from non-road equipment, including milling and excavation equipment, on a per-horsepower-operating-hour basis. Published equipment-specific loading capacities (e.g., from equipment manufacturer performance handbooks) could be used to estimate a conservative loading rate to project emissions on a mass-emission-per-mass-pavement-removed basis by using these emission factors as provided in NONROAD. This is a similar approach to that taken by the Washington State Department of Transportation in an LCA they undertook to compare PCC road rehabilitation alternatives (Weiland and Muench 2010). However, while this information would allow for estimating fuel-related air emissions due to equipment operation, no estimates of particulate matter emissions from road abrasion or environmental burdens associated with milling/excavation equipment manufacturing were found in the literature. Also, while it appears that water spray is used to suppress particulate emissions from milling, no estimates were found on the quantity or quality of the water released from spraying.

3. **Fuel-related and particulate matter emissions from RAP processing.** Only Wilburn and Goonan (1998) estimated the amount of energy necessary to process RAP. This energy was not categorized by fuel type (e.g. electricity, diesel). An estimate of the average quantity and mix of fuels used for RAP processing is necessary to develop a more accurate LCI dataset. Also, we found no source of data for particulate matter emissions released as a result of RAP processing. No information on the environmental burdens associated with the manufacture or decommissioning of RAP processing equipment was identified.

4. **Leachable emissions from the landfill disposal or use of RAP as aggregate.** Although an estimate of liquid emissions from RAP disposal in unlined and inert debris landfills and for use as aggregate is included in the proposed LCI datasets and was developed based on available batch and column RAP leaching test data, these estimates are probably lower than the actual emission due to partial leaching of contaminants attributed to the L:S ratio of these tests. Future research should consider estimating the long-term leaching of RAP.

Although the discussion presented in this chapter focuses on HMA, there has been significant growth in the use of WMA in recent years; approximately 12% of the asphalt pavement produced in 2010 was WMA (FHWA 2012), while approximately 24% of the asphalt pavement produced in 2012 was WMA (NAPA 2013a). While the percentages of asphalt and binder are generally the same for HMA and WMA, WMA production requires the use of an additive to reduce the viscosity of the asphalt so that it can be mixed and applied at a lower temperature than HMA. The resulting (28°C or more) temperature reduction impacts air emissions from the pavement production and application process (FHWA 2012). While RAP may be incorporated into WMA, it appears that standard practices for this beneficial use are not yet well established (TRB 2011). Development of LCI datasets for WMA production and EOL management should be considered as additional data on this technology become available.

Table 3-8. Overview of LCI Data Available

Process	WARM	GaBi	Athena	AP-42	Wilburn and Goonan (1998)	US LCI (2012)	Townsend and Brantley (1998)	Azah (2011)
Crude Oil Extraction	P	X	X			X		
Asphalt Production	P	X	X			X		
HMA Production	P			P				
Asphalt Pavement Removal								
Transport	P		X		P	X		
Landfill Disposal	P						P	P
RAP Processing					P			
RAP Use as Aggregate							P	P

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4 Asphalt Shingles

4.1 Introduction

Asphalt shingles are more commonly used over other roofing alternatives (e.g., wood, tile, slate, and metal) due to their relatively lower material and installation cost and their durability (ARMA 2014). Four out of five homes in the US are covered with asphalt shingles (ARMA 2014). More than 12 MMT (12.5 billion ft^2) of shingles are manufactured in the US annually; approximately 65% are used for re-roofing projects and 35% are for new roofs (Brock 2007). The sources of discarded shingles are post-manufacturing and post-consumer (i.e. from construction, renovation, and demolition activities). Post-consumer shingles generated from construction, renovation, and demolition activities are commonly referred to as *tear-offs*. Although old shingles may be overlain by new shingles during reroofing, most building codes limit maintenance of one reroof without removing the existing shingles. The shingles, therefore, are removed at some point after their service life, which typically is 20 years (NCHRP 2013).

Approximately 10 MMT of asphalt shingles are discarded annually in the US; in 2011, approximately 9 MMT were disposed of at permitted or registered solid waste disposal facilities (US EPA (2014)) and nearly 1.3 MMT were recovered for use in pavement production NAPA (2013). Roughly 90% of recovered asphalt shingles are comprised of tear-off shingle scrap and 10% represent post-manufacture scrap (VANR 1999, Sengoz and Topal 2005). Post-manufacture scrap tends to be more uniform (relative to tear-offs) and typically consists of shingles and packaging material (e.g., paper or plastic). However, tear-off shingles contain other roofing debris (e.g., wood, paper, metal, etc.) and have variable properties since different loads of shingles may have different asphalt composition and may have been subject to varying degrees of weathering or outdoor exposure.

The discarded shingles are transported either to a landfill for disposal or to a processing facility and eventually used for asphalt pavement production as depicted in Figure 4-1 While not an established practice in the US, the use of discarded shingles as a fuel source in an industrial application (e.g., cement manufacturing) has been explored on a limited scale (OCC 2008, Lee 2011). Figure 4-1 identifies the flow of materials and processes that should be considered for LCA of asphalt shingles EOL management.

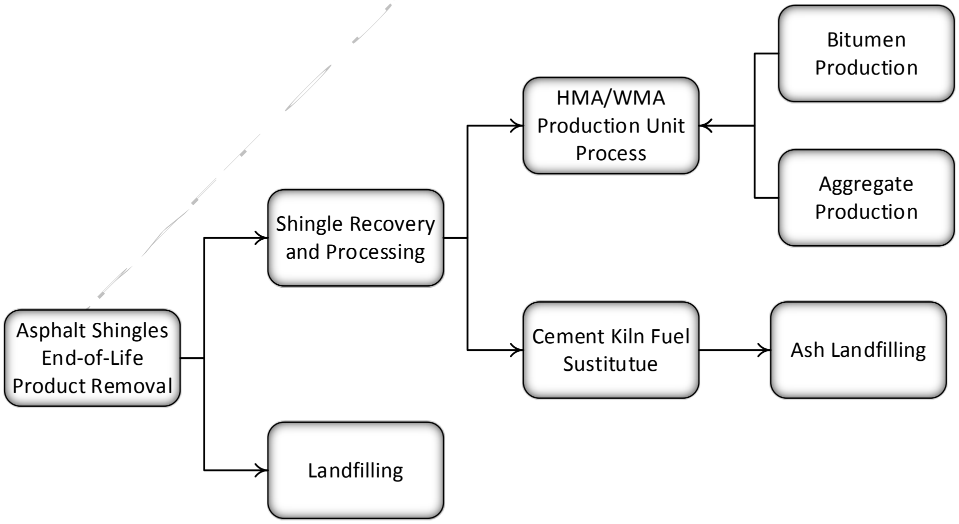

Figure 4-1. Asphalt Shingle EOL Management Processes

4.2 EOL Management

Several regional composition studies suggest that composition roofing (which consists of asphalt shingles and attached roofing tar and tar paper), comprises 2.7 to 18.3% (by weight) of CDD materials received at landfills (CCG 2006, CCG 2008, CCG 2009, CDM 2009, CDM 2010, and RWB et al. 2010). Based on the 88.1 MMT total amount of CDD materials disposed of at permitted or registered solid waste management facilities in 2011 (US EPA 2014), this suggests that approximately 2.4 to 16 MMT of composition roofing were placed in landfills.

As identified in Figure 4-1, three potential EOL management pathways for asphalt shingles are HMA production, combustion in a cement kiln, and disposal. Closed-loop recycling of asphalt shingles into new asphalt shingles is not a viable recycling option at present due to challenges in meeting stringent manufacturer feedstock specifications (Snyder 2001, OCC 2008). Consequently, applicable closed-loop recycling option and the associated LCI data (i.e., raw material extraction and product manufacturing data for asphalt shingles) are not discussed in this report.

Due to their substantial asphalt content (approximately 20% in fiberglass shingles), the use of asphalt shingles in asphalt paving mix production has increased significantly in recent years from 0.64 MMT in 2009 to 1.7 MMT in 2012 (NAPA 2013). In addition to asphalt, shingles also provide aggregate needed for paving mix production. Apart from paving mix production, paving mix producers recycled approximately 66,000 MT of asphalt shingles as aggregates in 2012 (NAPA 2013).

Asphalt shingles, due to their significant energy content, present a potential opportunity for energy recovery, including use as a supplemental fuel in cement kilns. Combustion of discarded asphalt shingles for energy is still under development in the US (OCC 2008, Lee 2011). The US EPA (2012) used emissions from the combustion of oil and lubricants as a proxy for the emissions from shingles combustion due to a lack of shingles-specific combustion emissions data and assumed that shingles combustion in cement kilns would offset emissions associated with refinery fuel gas combustion for assessing emission factors for the use of shingles in a cement kiln for the WARM model. As shingles combustion in a manufacturing application is not widely practiced, data needed to develop LCI for this process are lacking and shingles combustion is not further discussed in this report.

Using paving industry recycling estimates and the approximate amount of asphalt shingle waste produced annually, it is estimated that more than 80% of waste asphalt shingles are disposed of in landfills, typically within CDD landfills (Sengoz and Topal 2005, CIWMB 2007, CMRA 2007a, and NAPA 2013). Some landfill facilities may separate incoming loads of asphalt shingles for use as road base material for temporary access roads or for truck pads. Table 4-1 presents LCI needed to conduct an LCA of asphalt shingle EOL management.

Table 4-1. LCI Needed for LCA of Asphalt Shingles EOL Management

Process	Description
Asphalt Shingle Processing	The emissions associated with processing shingles for contaminant (e.g., nails) removal and size reduction include materials and energy input as well as process non-energy emission such as particulate matter emission from shingles grinding.
Landfilling	The material (e.g., equipment, soil, water) and energy (fuel, electricity) inputs for placement and compaction of discarded shingles in a landfill along with the associated process energy and non-energy emissions (e.g., dust emissions from equipment operation and liquids emission associated with physiochemical degradation of shingles in landfill) should be considered in developing a representative dataset for the landfill disposal of asphalt shingles.

Bitumen Production	The use of asphalt shingles for paving mix production displaces the production of primary bitumen. The material burdens, energy requirements, and process emissions related to the production of primary bitumen should be considered for an LCA of asphalt shingles used for paving mix production. LCI for this process are described in Chapter 3.
Aggregate Production	The use of asphalt shingles for paving mix production displaces the production and use of primary aggregate. The material burdens, energy requirements, and process emissions related to primary aggregate production should be considered for an LCA of asphalt shingles used for paving mix production. LCI for this process are described in Chapter 2.
Transportation	The emissions associated with transporting discarded asphalt shingles to recycling facilities or landfill, primary materials (for bitumen and aggregate production) to paving mix production plant, and processed shingles to respective end uses for the material should be included in the LCA.

4.3 LCI Sources

Peer-reviewed literature, government and private industry publications, and LCA modeling tools were reviewed to identify available LCI datasets pertaining to asphalt shingles EOL management processes. If LCI data were not available, process metadata and documentation were reviewed to evaluate the completeness of the dataset (e.g., emissions categories were included, background data were used to compile the dataset, and the geographic location and time period of the data were considered). The primary sources of information used to develop the LCI datasets and information identified, if available, were reviewed. Table 4-2 presents the data sources reviewed to compile LCI for shingle EOL management options.

Table 4-2. List of Sources Reviewed for LCI Data

LCI Source	Description
Athena (2000)	The report presents cradle-to-gate LCI for manufacturing various types of asphalt roofing products such as organic felt asphalt shingles and fiberglass mat asphalt shingles in Canada.
Athena (2001)	Franklin and Associates developed LCI for Road and Roofing Asphalt, which includes emissions associated with crude oil extraction and processing at a petroleum refinery for asphalt production (i.e., cradle to gate) in the US.
US LCI (2012)	The National Renewable Energy Laboratory has published LCI for a wide variety of materials, products, and processes used in the US.
Cochran (2006)	Cochran (2006) presented diesel energy requirements for asphalt shingles processing.
Trumbore et al. (2005)	Trumbore et al. (2005) presented air emissions based on measurements from more than 20 asphalt roofing manufacturing facilities and one pilot plant in the US. Emission factors were developed using these measurements and proposed to update the older US EPA AP-42 asphalt roofing manufacturing emission factors.
US EPA (2012)	EPA’s Waste Reduction Model presents data on GHG emissions associated with source reduction, transport, recycling, and landfilling (i.e., collection and placement) of asphalt shingles.

LCI Source	Description
AP-42 (US EPA 1995)	Provides emission factors of filterable particulate matter (PM), total organic carbon (TOC), and carbon monoxide (CO) for asphalt roofing manufacturing processes.
NIST (2011) BEES	The National Institute of Standards and Technology Building for Environmental and Economic Sustainability model allows an economic and environmental impact comparison between various building materials. Documentation suggests that the model uses data from SimaPro, US LCI (2012) and Trumbore et al. (2005) for estimating LCI for asphalt shingles manufacturing. No LCI specific to EOLmanagement of shingles are provided.
GaBi (PE International n.d.)	GaBi includes US-specific LCI for crude oil, aggregate (i.e., limestone), and asphalt production; the database does not contain asphalt-shingles-specific LCI.
Ecoinvent (2014)	Ecoinvent is an LCI database developed by the Swiss Centre for Life Cycle Inventories, which includes specific processes related to the EOL management of numerous individual materials. Asphalt shingles are not addressed by Ecoinvent.

4.4 Asphalt Shingles Manufacturing

As discarded asphalt shingles are not used for manufacturing new asphalt shingles (closed-loop recycling), asphalt shingles manufacturing LCI do not impact the EOL management of discarded shingles. This section provides a brief overview of shingle manufacturing and references to available LCI data relevant to this process.

Asphalt shingles are typically comprised of four main constituents: aggregates, asphalt, an asphalt-impregnated mat, and a fine mineral base. The six major sequential operations used for asphalt shingle manufacturing are felt saturation, coating, mineral surfacing (top and bottom), cooling and drying, product finishing, and packaging. The shingle fabrication process begins with coating a layer of organic (cellulose or wood fiber) or fiberglass fiber mat with asphalt by passing it through a tank filled with hot “blown” asphalt. Asphalt, prior to use in shingles manufacturing, is oxidized by bubbling oxygen into the liquid asphalt until the desired properties (e.g., viscosity) are achieved; this process is referred to as “blowing” (NIOSH 2001, Blachford and Gale 2002, and Wess et al. 2004).

The mat material supports the other components while the asphalt provides weather resistance and waterproofing. The organic or fiberglass mat makes up 2-15% of the shingle and the asphalt binder comprises from 19-36% of the mass of an asphalt shingle. Fiberglass and organic felt are the two types of mats used for asphalt shingles manufacturing in the US. Organic felt is made of cellulose fibers, while the fiberglass mat is generally made by chopping fine glass filaments and mixing them with water to form a pulp, which is then formed into a sheet (Blachford and Gale 2002). Fiberglass asphalt shingles are most commonly used shingle type in the US (Athena 2000).

Once coated with the appropriate thickness of asphalt, one side of the mat is then surfaced with granules. The surface granules consist of crushed rock coated with ceramic metal oxides for protection against physical damage and sun-exposure. The granules add a desired color to the product and may also contain a chemical such as copper to inhibit algae growth during the shingle’s service life (3M 2014). Granular aggregates comprise from 20-38 % of the weight of an asphalt shingle. A light coating of fine sand, talc, or fine particles of mica is applied to the back surface of the shingle, which represents the bottom surface of the shingles, to prevent the individual shingles from adhering to each other during packaging and transport (Blachford and Gale 2002; Grodinsky et al. 2002; Willett 2013). Mineral fillers comprise 8-40 %

(by mass) of an asphalt shingle. The final steps in the production of asphalt shingle are the finish, cutting, and packaging of the shingles (NIOSH 2001, CMRA 2007a).

Asphalt shingle manufacturing LCI data are available through several sources. Athena (2000) presents materials and energy use, and air, water and solid waste emissions from the asphalt shingle manufacturing process based on data from facilities in Canada. AP 42 presents filterable particulate matter, total organic carbon, and carbon monoxide emissions factors developed for a range of different asphalt blowing and shingle saturation processes based on air pollution emissions tests from five asphalt roofing manufacturers in the US during the 1970s. Due to limited or poor test data, these emission factors are rated as below average (D) to poor (E) by the US EPA. Trumbore et al. (2005) proposed emissions factors for particulate matter, volatile organic compounds, sulfur oxides, carbon monoxide, nitrogen oxide, and 22 hazardous air pollutants based on measurements from more than 20 shingles manufacturing facilities in the US. As the discarded asphalt shingles are not currently recycled to produce new shingles, the emissions and LCI data presented by these sources were not further evaluated and considered as these upstream processes do not impact EOL management.

4.5 LCI Related to Disposal

Emissions associated with shingles disposal in a landfill include air emissions from equipment used for placing shingles in the landfill, emissions associated with landfill construction and operation, and liquids and gaseous emissions from material decomposition in the landfill environment. Asphalt shingles are not expected to significantly degrade biologically and, therefore, not expected to produce gaseous emissions (US EPA 2012). There are various sources which provide landfilling emission factors related to equipment use and landfill construction and operation; however none are specific to asphalt shingle landfilling. Generalized landfill construction and operations LCI data were presented in Chapter 2.

Leaching may be a potential environmental concern with asphalt shingles because asphalt products contain PAHs (Kriech et al. 2002, CMRA 2007a). Asphalt shingles are typically disposed of with other discarded CDD materials and not in a monofill; field-scale leachate-quality data specific to asphalt shingles disposal in a landfill are not available. The available laboratory-scale shingles leaching studies data were reviewed for developing an estimate of liquids emission from shingles disposal in landfills. Kriech et al. (2002) measured the total and leachable (leached using a TCLP solution) concentration of 29 PAHs in four primary roofing asphalt samples from a commercial source of roofing asphalt for built-up roofs. The total PAH results indicated concentrations in the roofing shingles ranging from 4.0 to 23 mg/kg. None of the 29 PAHs analyzed were detected in TCLP leachate, suggesting that PAHs do not readily leach from different asphalt materials.

Commercial Recycling Systems (Scarborough, Maine) reported leaching (TCLP) results for ground shingles (CDRA 2010). The results indicated that the VOCs, Semi-VOCs, PAHs, and metals were not readily leachable. Low concentrations of some metals (eight RCRA metals) were reported. Some constituents of ground shingles, most notably cPAHs, were reported to exceed the concentration standards for state *de minimis* risk levels (Appendix A of Chapter 418 Maine Solid Waste Rules [MDEP 2012]). The data were not available for further evaluation.

Azah (2011) conducted batch and column leaching tests (SPLP) on a shingle sample collected from a recycling facility in Florida to assess PAHs leaching from shingles. These data were used to estimate PAHs emissions from disposal of shingles in CDD materials in landfills. Batch test data were used for PAHs that were measured above the method detection limits. Batch test concentrations were multiplied by the total solution volume and divided by the sample mass to estimate leachability on a per-kilogram-asphalt-pavement basis (Table 4-3). The column leaching test (under saturated conditions) data were used to estimate leaching emission of PAHs (acenaphthene, phenanthrene, anthracene, and dibenzo(a,h)anthracene

) that were not detected in batch leaching tests. The column leaching test was conducted under saturated conditions until a L:S ratio of 3.08 (L of liquids per kg of shingles) was achieved. PAHs concentrations were measured at liquids-to-solid ratios (0.62, 1.23, 1.85, 2.47, and 3.08 L of liquid per kg of shingles). The contaminant mass released between two sampling events was estimated by multiplying the L:S ratio increment from the previous sampling event to measured PAH concentration. The cumulative leaching amount was estimated by adding the leaching amount from each sampling interval.

Jang (2000) conducted batch and column leaching tests of several individual CDD materials, including asphalt shingles using the SPLP extraction fluid to assess leaching of conventional water-quality parameters. These data were used to estimate liquids emissions of calcium, chloride, and sodium with asphalt shingles disposal in an unlined inert materials landfill; the parameter measured below detection were not used. The concentrations were multiplied by the total solution volume and divided by the sample mass to estimate leachable mass per kilogram of shingles (Table 4-3). The results for nitrate and sulfate were not included due to the presence of nitric acid and sulfuric acid in the SPLP leaching solution, the influences of which are not entirely known.

The asphalt shingles leaching data and energy consumption data from landfill operations were used to develop an LCI process dataset for disposal of asphalt shingles at an unlined CDD landfill, as presented in Table 4-3. Emissions are provided per kilogram “Asphalt shingles, at unlined CDD landfill” flow. Although the actual liquids emissions are expected to be greater than the estimated liquids emission, as the material would be subjected to leaching a higher L:S ratio than that used by Azah (2011) and Jang (2000) for the batch leaching tests, using these emissions for LCA until the total emission estimates become available would be more accurate than excluding liquids emission altogether. Details on diesel and electricity consumption as a result of landfill operations (included in the “CDD landfill operations” flow) and details on the calculation for cover soil requirements for placement of asphalt shingles at an unlined CDD landfill are provided in Chapter 2. Tthe bulk density of asphalt shingles provided by CCG (2006) was used to estimate cover soil requirements. In the absence of average nationwide distance data, the site of asphalt shingle removal was assumed to be 20 km from the CDD landfill disposal site.

Table 4-3. Proposed LCI Dataset: Asphalt Shingles, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
Asphalt shingles, from roof removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.0438
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Asphalt shingles, at unlined CDD landfill			*kg*	*1*
Acenaphthene	Azah (2011)	Water/Groundwater	mg	0.00171
Phenanthrene	Azah (2011)	Water/Groundwater	mg	1.03E-05
Anthracene	Azah (2011)	Water/Groundwater	mg	0.00210

Fluoranthene	Azah (2011)	Water/Groundwater	mg	0.0147
Pyrene	Azah (2011)	Water/Groundwater	mg	0.011
Benzo(k)fluoranthene	Azah (2011)	Water/Groundwater	mg	0.00102
Chrysene	Azah (2011)	Water/Groundwater	mg	0.00135
Benzo(g,h,i)perylene	Azah (2011)	Water/Groundwater	mg	0.00673
Benzo(a)pyrene	Azah (2011)	Water/Groundwater	mg	0.00333
Benzo(a)anthracene	Azah (2011)	Water/Groundwater	mg	0.0065
Dibenzo(a,h)anthracene	Azah (2011)	Water/Groundwater	mg	8.6E-06
Benzo(b)fluoranthene	Azah (2011)	Water/Groundwater	mg	0.00207
Calcium	Jang (2000)	Water/Groundwater	mg	102
Chloride	Jang (2000)	Water/Groundwater	mg	146
Sodium	Jang (2000)	Water/Groundwater	mg	108

4.6 LCI Related to Recycling

4.6.1 Shingle Processing

Asphalt shingle processing includes removing contaminants (e.g., nails, metal flashing, plywood) from discarded asphalt shingles and reducing their size. The emissions and energy requirements for processing equipment should be considered for asphalt shingle processing LCI. Asphalt shingles, once removed, may be segregated from other CDD materials at the construction site or they may be commingled with other demolition waste. While some asphalt shingle processing facilities receive only source-separated asphalt shingles, others may not require source separation and perform all sorting operations to segregate asphalt shingle from other materials. Source separation places the burden on the construction/demolition contractor to remove non-shingle materials from the discarded shingles.

The degree and methods of processing asphalt shingles depend on the quality of materials received and facility design; post-consumer shingles require more intensive sorting due to the higher level of contamination compared with post-manufacture scrap shingles;. Post-consumer asphalt shingles may also have variable properties due to variation in degree of weathering and age among loads of asphalt shingles (NAHB 1998). Asphalt shingles that arrive at a mixed CDD facility are typically manually picked from the CDD debris and stockpiled until an appreciable amount of material has been acquired for processing. Once the shingles have been sorted and all the undesirable materials removed, the shingles are size-reduced using various grinding and screening methods to obtain the size necessary for the intended recycling application (CMRA 2007b).

Various grinding and screening methods have been used to grind shingles for recycling, including shredders, hammer mills, and different screen arrangements. A grinder typically consists of a loading hopper, feeding drum, grinding chamber, size screen, and an exit conveyer. Water is sometimes added during shredding to keep the grinder and shingles cool and to control dust (CMRA 2007b, TRB 2013). The ground-up shingles are typically screened. The fraction that does not pass through the screen may be used for a process with larger size specifications or they may be fed back into the grinder for further size reduction (Marks and Gerald 1997, VANR 1999, and Grodinsky et al. 2002). Screenings that are greater than ¾ of an inch can typically be used as an aggregate; in most HMA applications the shingles must be reduced to a size smaller than ½ an inch (CMRA 2007b). Sand may be added to the ground shingles to prevent agglomeration of the materials during storage (IWCS 2010, TRB 2013).

The typical fraction of recovered shingles for an asphalt shingles processing facility (not requiring source separation) ranges from 15 to 90%. The recovery rate is dependent on the quality of the feedstock shingles and the efficiency of the sorting equipment. Facilities achieving the highest recovery rates are those that target loads comprised mostly of just asphalt shingle waste; facilities that accept and sort mixed CDD materials would have much lower asphalt shingle recovery rates (CMRA 2007b).

Particulate matter generation from the grinding process and the potential release of asbestos are potential non-energy air emission sources from asphalt shingles processing. Similarly, liquids emissions to surface and groundwater from short-term storage of unprocessed and processed shingles in stockpile should be considered for quantifying LCI. Asbestos, a known health hazard, was used in roofing products up until the 1970s. Although asbestos is not used as frequently today in roofing products (e.g., cements, mastics and other products may still contain asbestos), post-consumer shingle waste removed from older housing may potentially contain residual asbestos from when asbestos products were more commonly used (NAHB 1998, USGS 2010). An important goal for CDD recycling facilities when processing asphalt shingles is to ensure compliance with asbestos regulations (typically accomplished by following an approved sampling protocol) and to minimize processing asphalt shingles containing asbestos (CMRA 2007a). One practice that is employed at CDD recycling facilities in states that require asbestos testing on asphalt shingles is to have a staging area where incoming roofing waste loads are held while asbestos analytical results are obtained (CMRA 2007a). Once analytical results indicate that the shingles do not contain asbestos, they are then moved from the staging area to the processing area; asbestos-containing shingles are diverted to disposal.

Based on an evaluation of approximately 28,000 shingles samples analyzed for asbestos content, CMRA (2007a) reported that 0.06% and 1.46% of samples has asbestos detected as less than 1% and more than 1% of asbestos content, respectively. These measurements quantify the total asbestos content of shingles and not the fraction that would be released into the air with grinding and screening. The asbestos emissions and amount of particulate matter, in general, from shingles processing and liquids emission from short-term storage of processed and unprocessed shingles are not available.

Energy inputs and emissions associated with asphalt shingle processing include those related to the manufacturing and use of sorting, grinding, and screening equipment. For shingles that are processed for use in HMA applications, Cochran (2006) reported diesel equipment energy requirements of 41 MJ per MT of asphalt shingle processed, which is equivalent to the combustion of 1.06 L of diesel per MT of shingles. US EPA (2012) used the energy requirement reported by Cochran (2006) to estimate shingles processing emission factors for WARM. Table 4-4Table 4-4 presents process energy requirements for shingles grinding based on the total diesel consumption estimated for these processing equipment data reported by Cochran (2006). It was assumed that the shingles will be transported 20 km from the point of generation to the processing plant. The energy requirement does not include energy needed to segregate shingles from mixed CDD materials. Process non-energy emissions (e.g., particulate matter emission and asbestos emission to air and leachate emission to surface and groundwater from asphalt shingles stockpile) and the emissions from manufacturing, maintaining, or disposing of/dismantling the grinding equipment are not included due to a lack of available information. Grinding and screening of post-consumer shingles also produces ferrous metals (nails) (IWCS 2010). IWCS (2010) visually estimated separation of approximately 2.5 cubic feet of nails with the grinding and screening of 90 MMT of post-consumer asphalt shingles. As the mass of the nails was not measured by IWCS (2010), the data were not used for the LCI presented in Table 4-4.

Table 4-4. Proposed LCI Dataset: Asphalt Shingles, at Processing Plant

Input Flow	Source	Category	Unit	Amount
Asphalt shingles		Construction and Demolition Debris Management	kg	1.0
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Diesel, combusted in industrial equipment	Cochran (2006)	Flows	L	0.00106
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Ground asphalt shingles, at processing plant		*Construction and Demolition Debris Management*	*kg*	*1.0*

4.6.2 HMA/WMA Production with Asphalt Shingles

Use of asphalt shingles for producing paving mix has significantly increased in recent years due to the shingles' asphalt content and the increase in asphalt prices. The benefits of using asphalt shingles for paving mix production include reduced demand on primary asphalt cement and aggregate, reduced paving mix production cost, and improved resistance to pavement cracking and rutting due to the reinforcement provided by fibers contained in shingles (Brock and Shaw 1989, Grzybowski 1993, Ali et al. 1995, Button et al. 1995, NAHB 1998, Foo et al. 1999, Mallick 2000, and Sengoz and Topal 2005). Several state departments of transportation (e.g., Georgia, Minnesota, Montana, and South Carolina) have specifications allowing from 3 to 8% of HMA to be replaced by tear-off shingles; other laboratory experiments have acknowledged that up to 7% of HMA material can be replaced by tear-off shingles without adverse effects (Mallick 2000, OCC 2008).

Another potential environmental concern, apart from asbestos release from shingle grinding, with the recycling of asphalt shingles is the emission of PAHs during the production of HMA. Since asphalt is a mixture of paraffinic and aromatic hydrocarbons, heating of asphalt can result in the emission of PAHs (ARMA 1998; US EPA 2000; Lee et al. 2004). PAHs are one of the major classes of air pollutants emitted from HMA facilities (US EPA 2000). While the quantity of PAH emissions from HMA facilities has been fairly well documented, the impact of using recycled asphalt shingles on PAH emissions is not well understood. Currently, there are no available data to suggest that emissions of PAHs during HMA production with recycled asphalt shingles would be different from emissions from production without asphalt shingles. TRB (2013) reported that HMA plants may require more frequent cleaning and adjustments may need to be made to temperature settings to melt the more hardened shingle asphalt, suggesting greater energy demand for HMA production using asphalt shingles. However, data are not available to estimate the additional energy and material demand associated with using shingles for HMA production.

Asphalt and aggregates constitute approximately 19-36% and 20-38% (by weight) of shingles, respectively (NAHB 1998, CIWMB 2007). Therefore, asphalt shingles supplements the use of primary asphalt and aggregates in paving mix production. The US EPA (2012) assumed that asphalt shingles provide 22% and 38% (by weight of shingles) of asphalt and aggregate, respectively, for estimating emission offsets associated with avoiding primary asphalt and aggregate production. The US EPA (2012) also assumed a loss of 7.2% of the product during recycling. Based on the procedure used by the US EPA (2012), recycling 1 kg of asphalt shingles for HMA production is estimated to replace 0.2 kg and 0.35 kg of primary asphalt and aggregates, respectively. It is assumed that there are no differences in emissions associated with use of asphalt shingles in HMA production (e.g., particulate matter, liquid, and the energy requirements) as compared to those from the use of primary materials in the production process. It is assumed that there would be additional avoided emissions resulting from the prevention of the transport of primary aggregates and asphalt to the HMA plant; additional details on the transport distances for these primary materials are provided in Chapters 2 and 3.

US EPA (2013) conducted LCA of recycling asphalt shingles into asphalt pavement. While the report discussed the collection of energy data from shingles processing facilities, these data are not provided in the report and could not be used for the development of an LCI specific to shingle processing for recycling in asphalt pavement application.

4.7 Data Gaps and Future Opportunities

Table 4-5 summarizes the type of data presented by various sources reviewed for compilation of asphalt shingle EOL management LCI. Only WARM (US EPA 2012), Athena (2001), AP-42 (US EPA 1995), MSW-DST, Azah (2011), Wilburn and Goonan (1998), and GaBi provide information with respect to US-based processes. Some sources used data from the other sources presented in Table 4-5. For example, the US EPA (2012) used data from NREL database and Cochran (2006). As shown in the table, many sources present only part of the data/information needed for compiling LCI. For example, WARM uses only GHG emissions associated with equipment fuel consumption to estimate landfill emission. Similarly, Ecoinvent only has partial landfill leachate emissions data because leachate from inert materials landfills are not considered.

A majority of LCI information available on asphalt shingles pertains to the manufacturing aspects of the life cycle. Only limited EOL-specific LCI are available. Based on a review of the available information, the following data gaps were identified for compilation of a more comprehensive LCI dataset for asphalt shingles EOL management:

1. **Long-term leachable emissions from asphalt shingles placed in a landfill.** As described earlier, the liquid emissions presented in this study are based on SPLP tests, which simulate leaching from land-application or disposal in inert debris landfills. Although references to results of batch test (TCLP) data mimicking leaching from shingles disposed of in MSW landfills are found (CDRA 2010), the actual data could not be located for developing liquids emissions estimates pertaining to shingles disposal in MSW landfills. Moreover, the batch leaching data used for estimating liquid emissions correspond to liquid to a solid ratio of 20 and, therefore, do not represent complete liquid emission. As asphalt shingles are typically disposed of with other discarded materials and not disposed of in a monofill, field-scale leachate quality data specific to asphalt shingles disposal in landfill are not available and probably will not be available in the future. The liquids emissions from asphalt shingles placement in an inert materials as well as MSW landfills would, therefore, need to be based on laboratory-scale studies simulating long-term liquids emissions. Leaching studies have been published on asphalt shingles as a component in the CDD debris waste stream and on asphalt binder used for built-up roofs, which are a type of asphalt roofing but different from asphalt shingles (Townsend and Kibert 1998, Townsend et al. 1999, Kriech et al. 2002, and Jang and Townsend 2003).

   Leaching or air emissions data have been collected by facility operators and submitted to the state environmental agencies for compliance with state rules and present the opportunity to develop asphalt shingle leaching LCI based on actual operations. For example, Commercial Recycling Systems (Scarborough, Maine) reported asphalt shingles leaching (TCLP) results and asbestos analysis to the Maine Department of Environmental Protection for permitting of an asphalt shingles processing facility.

2. **Materials and energy input and emission from asphalt shingles processing.** The shingles processing LCI presented in this report are based on the energy requirements reported by one data source (Cochran 2006). Cochran (2006) identified the energy requirements of shingles processing based on a survey of a handful of equipment manufacturers. Some of the data (e.g., consumables, fuel and electricity usage, water consumption, material throughput) tracked by the facility owner from financial accounting perspective can readily be used for developing more comprehensive LCI for shingle processing.

The process non-energy emission (e.g., particulate matter and asbestos emissions to the atmosphere) associated with asphalt grinding are lacking. Future research should focus on collecting and compiling these data.

3. **Data pertaining to shingles use in pavement mix production**. The use of asphalt shingles in HMA production has shown a significant increase in recent years. Similar to the US EPA (2012), the LCI presented in the report for shingles use for HMA production are based on the assumption that shingles do not impact emissions associated with HMA production. Data are not available to estimate the additional energy and material demand and emissions associated with using shingles for HMA production. Future research should consider assessing the impact of shingles on energy and materials input and emissions associated with HMA production. The impact of using shingles in HMA on the quality and service life of the pavement should also be assessed. The discussion presented in this chapter focused on recycling asphalt shingles in HMA. However, there has been significant growth in the use of WMA in recent years. There are insufficient data pertaining to the use of shingles in WMA.

4. **Data pertaining to asphalt shingles use as fuel source in industrial applications.** The LCI data for asphalt shingles use as a supplemental fuel in cement kiln are lacking due to the rarity of this practice in the US (OCC 2008). The cement industry has experimented with the use of asphalt shingles in cement kilns [e.g., Lafarge cement plant, Brookfield, Nova Scotia, and St. Mary's Cement, Charlevoix, Michigan (Lee 2011)]; however, there is very little research data available on the subject. A Department of Energy-sponsored project conducted in 2007 investigated the feasibility of using asphalt shingles (post-manufacture and post-consumer) in the manufacture of cement and in circulating fluidized bed boilers (OCC 2008). OCC (2008) presented emissions from the combustion of the shingles and the potential impacts on the quality of the products and the cement kiln dust (CKD); however, the data were interpreted as being rudimentary. None of the existing US-specific data sources listed in Table 4-5 except WARM (US EPA 2012) included emissions associated with shingles in cement kiln manufacturing. The WARM model does consider GHG associate with combusting shingles in a cement kiln; however, GHG were estimated using lubricants as a proxy to estimate emissions from the combustion of fiberglass asphalt shingles. Future research into this potential recycling application is recommended.

Table 4-5. Overview of LCI Data Available

Process	**WARM**	**MSW-DST**	**Ecoinvent**	**GaBi**	**Athena (2001)**	**AP-42**	**US LCI (2012)**	**Wilburn and Goonan (1998)**	**Jang (2000)**	**Azah (2011)**	**Cochran (2006)**
Aggregate Production	P		X	X		P	X	P			
Asphalt Production	P			X	X	P	X				
Transport	P				X			P			
Landfill Construction & Operation	P	X		X							
Landfill Leachate Emissions			P						P	P	
Shingle Processing	P										P
HMA Production	P										
Shingles in Cement Manufacturing											

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5 Gypsum Drywall

5.1 Introduction

Gypsum drywall (also referred to as gypsum board, wallboard, or plasterboard), typically manufactured and sold as 4-ft-by-8-ft or 10-ft-by-12-ft sheets or panels, is widely used as an interior wall and ceiling finishing in residential, commercial, and institutional structures. Although there are a variety of drywall products, including many varieties of fire-resistant and water-resistant product, ½-inch-thick regular and Type X (fire-resistant) gypsum boards combined constitute over 80% (by weight) of the prefabricated gypsum products (Crangle 2014). In 2013, approximately 19.5 million ft^2 of gypsum drywall products were sold in the US (Crangle 2014).

Primary mined gypsum, flue gas desulfurization (FGD) byproduct gypsum, and recycled gypsum are all input streams used for gypsum drywall manufacture. In 2013, approximately 90% of the 24 MMT of gypsum consumed in the US was used by manufacturers of drywall and plaster products (Crangle 2014). In combination with other additives, a gypsum slurry is made that is deposited on paper facing and backing to create sheets of drywall. These sheets are dried and cut to specific sizes and distributed for use in construction and renovation projects. Once delivered to the jobsite, drywall sheets are attached to the interior frames of the building using drywall screws and are cut to fit the various structures in the building. Drywall sheet-fitting generates scraps that are often free of tarnish or paint. Figure 5-1 presents the various gypsum drywall processes, including production and EOL management; processes upstream of the EOL removal are depicted because some of the post-consumer gypsum is recovered and used in lieu of mined gypsum (i.e., recycling of waste drywall to new drywall or agricultural use). Such uses of the recovered drywall results in energy and emission offsets associated with avoidance of gypsum mining and production.

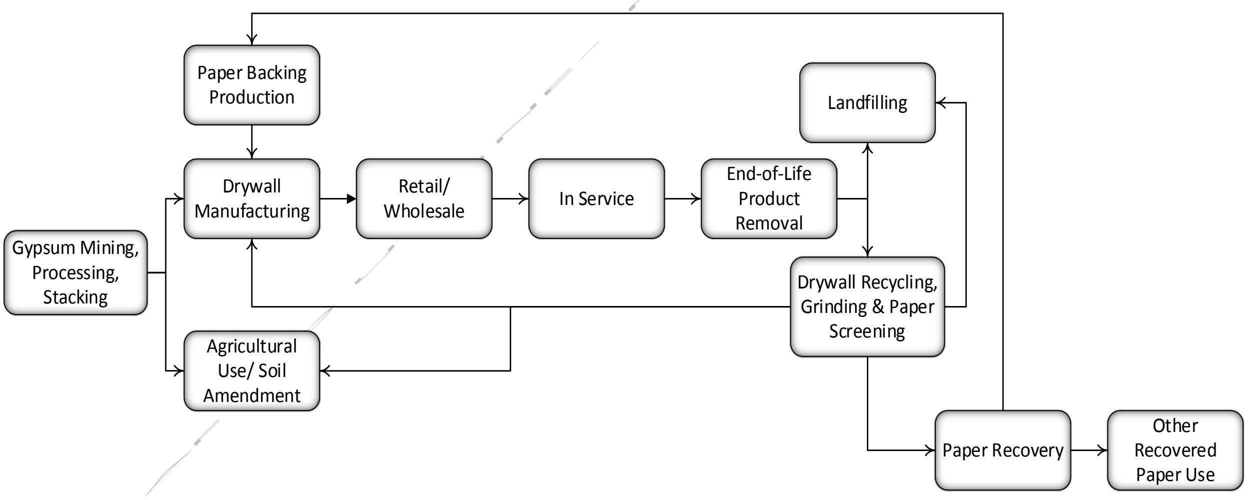

Figure 5-1. Gypsum Drywall Process Flow Diagram

5.2 EOL Management

US EPA (2014) estimated that nearly 7.4 MMT of waste gypsum drywall was handled by permitted or registered solid waste management facilities in the US in 2011; approximately 6.9 MMT was disposed of in C&D and MSW landfills and about 0.5 was recovered through CDD processing facilities. CIWMB (2007) reports the fractions of (California) waste drywall which are produced from new construction, demolition, manufacturing, and renovation activities; these fractions are presented in Figure 5-2. Distinguishing between these waste drywall sources is important because the drywall quality is substantially different among these streams; manufacturing and construction waste drywall are relatively clean, while demolition and renovation drywall typically has contaminants such as paint, joint compounds, nails, and any other material that may have been applied to the drywall during installation and use.

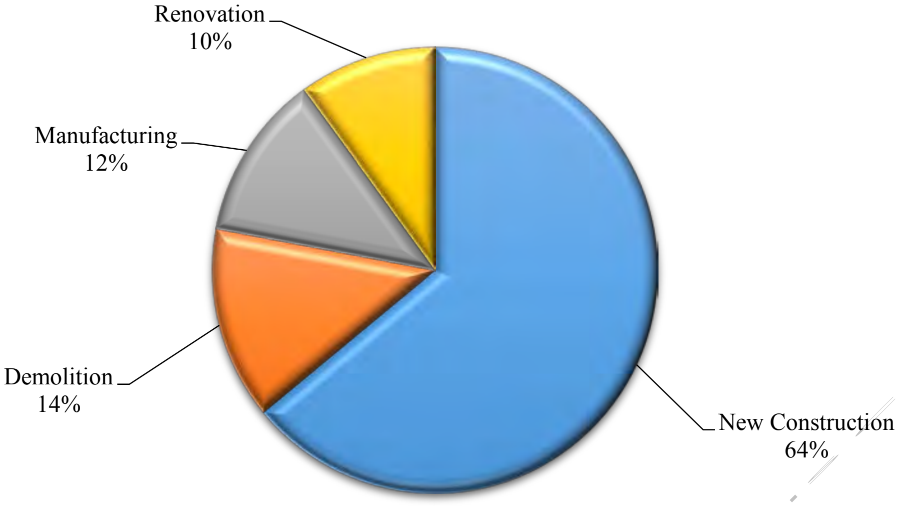

Figure 5-2. Sources of California-Discarded Gypsum Drywall (CIWMB 2007)

Due to its relatively high quality (e.g., absence of nails, paint), the drywall discarded from drywall manufacturing and new construction activities accounts for the bulk of the drywall waste stream recycled (Venta 1997, WRAP 2008, US EPA 2012). Gypsum drywall that is not recycled (representing the majority of waste drywall) is disposed of in landfills. Drywall is estimated to account for approximately 8% of the landfilled CDD materials based on several regional waste composition studies (Barnes 2000, Sandler 2003, CCG 2006, CDM 2009, RWB et al. 2010). MSW landfills may have restrictions on the amount of drywall material that is disposed of and where it is disposed of, or it may be banned from disposal altogether (e.g., Massachusetts) often due to anticipated or actual problems related to the generation and release of hydrogen sulfide gas. However, it appears that the majority of demolition drywall is still landfilled. RSM may contain significant amounts of gypsum and may be used as daily cover in landfills. The fate and the associated environmental impacts of gypsum in a landfill in either case (disposed of as waste or used as daily covers) would, potentially, be the same.

Drywall which is recovered for recycling is typically taken to drywall processing facilities where processing removes contaminants and separates gypsum from the paper backing (WRAP, n.d.). The processed drywall can be recycled in closed loop (e.g., new drywall manufacturing) or open loop (e.g., soil amendment) application. Closed-loop recycling and some open-loop recycling applications result in avoiding primary gypsum mining and production and associated emissions. Drywall scrap recovered from construction sites can be ground and reused in a variety of agricultural purposes, such as a soil conditioner and liming agent (Marvin 2000). The recycled gypsum is applied directly to the ground in this application; reuse permits or regulations, if applicable, may dictate the application rate (e.g., allowable application, typically expressed in tons per acre per year). Concerns with drywall land application include dust and liquids emissions from processing and spreading gypsum drywall (Marvin 2000). Open-loop recycling of gypsum drywall may also include use as an additive to compost, acting as a bulking agent. The gypsum absorbs moisture in the pile and adds calcium, sulfur, and some carbon to the compost (Marvin 2000).

Table 5-1 presents processes that should be considered for LCA of EOL management options for gypsum drywall. For all the applications, where discarded drywall is processed to produce gypsum (referred to herein as *recycled gypsum*) and used as a substitute for primary gypsum, production of primary gypsum was identified as an end point for LCI compilation. The production and use of the recycled gypsum potentially would result in avoiding an equivalent amount of primary gypsum production and the associated emissions. Emissions downstream of the primary gypsum production (e.g., those associated with drywall manufacturing from primary gypsum), however, were assumed to occur irrespective of whether downstream processes used primary or recycled gypsum unless the available data indicated otherwise. As available, LCIs were developed for the processes listed in Table 5-1.

Table 5-1. LCI Needed for LCA of Gypsum Drywall EOL Management

Process	Description
Gypsum Mining and Production	The process entails mining and processing (size reduction via grinding and crushing) of primary gypsum for use in drywall manufacturing, cement production, or agricultural use.
Paper Backing Production	The process includes manufacturing paper backing using current mix of primary and recycled sources for eventual use as facing and backing for gypsum drywall.
Drywall Manufacturing	The process entails processing and calcining gypsum to produce stucco, stucco slurry production, and forming, cutting, and drying gypsum drywall.
Landfilling	The process entails placing and compacting drywall and long-term physical, chemical, and biological decomposition of drywall in a landfill.
Drywall Grinding and Paper Screening	Processing includes grinding waste drywall and screening paper and other contaminants that may be present in the drywall waste to produce ground-up gypsum.
Paper Recovery and Recycling	The process includes recovering paper for reuse as input for new paper backing production for use in new drywall manufacturing or other uses.
Transportation	The fuel requirements and emissions from transporting primary materials and discarded drywall to recycling facilities and respective end uses for the material or to landfill should be considered in the LCA.

5.3 LCI Sources

Peer-reviewed literature, government and private industry publications, and various LCA modeling tools were reviewed to identify available LCI datasets pertaining to gypsum drywall EOL management processes. Table 5-2 lists data sources reviewed to compile LCI presented in this chapter. If LCI data were not available, process metadata and documents were reviewed to evaluate the completeness of the dataset (e.g., emissions categories, background data used to compile the dataset, and geographic location and time period of the data were included). The primary sources of information used to develop the LCI datasets and information identified, if available, were reviewed.

Although gypsum management LCI data were located in many of the sources listed below (Venta (1997), WRAP (2008), Ecoinvent, WRATE, GaBi), non-US-specific data are presented and discussed for a better understanding of the inputs used to develop LCIs for this material.

Table 5-2. List of Sources Reviewed for LCI Data

LCI Source	Description
Athena (2011)	Athena Institute developed this report for the Gypsum Association. The report presents cradle-to-gate LCI of 1/2-inch regular and 5/8-inch Type X gypsum drywall. It also presents US-specific LCI for raw gypsum extraction, gypsum paper manufacture, and drywall assembly based on the data from seven quarries/mining sites, three paper plants, and 17 drywall board plants to develop the LCI datasets with the intent of uploading them to the US LCI (2012) database.
Venta (1997)	The report presents cradle-to-gate LCI of different types of gypsum boards and associated finishing products. These LCI appear to be used by the Athena Impact Estimator for Buildings, an LCA model which evaluates the environmental impacts for structures created from a variety of building materials. The data used for LCI development were primarily specific to Canada. US-specific data such as distribution of different type of boards produced were used for developing the LCI.
EPA WARM	The US EPA's WARM presents energy and GHG emissions associated with source reduction, recycling, and landfilling of drywall. The data presented by Venta (1997) were used for developing emission factors for raw material extraction and drywall and paper manufacturing. The energy requirements provided by WRAP (2008) were used for developing emission factors for processing discarded drywall.
Cochran (2006)	Cochran provided an estimate of the energy requirements of equipment for processing gypsum drywall for recycling (in MJ/hour) based on a survey of equipment manufacturers.
Ecoinvent	This Swiss-based LCI database presents industrial LCA and management data. The database contains individual international gypsum production/management processes, including gypsum board production, gypsum quarry operation, recycling of waste gypsum board, and disposal of waste gypsum at sanitary and inert debris landfills.
WRAP (2008)	This report presents the results of an LCA, including the LCI used, of Type A plasterboard in the United Kingdom. Data considered are representative of industry practices and waste management for drywall (referred to as plasterboard) in the United Kingdom. Individual LCI datasets were developed using a combination of data from Ecoinvent, UK gypsum industry data, and primary information from manufacturing facilities.
GaBi	This LCA software contains its own database developed for specific processes similar to the Ecoinvent database. Data related to internationally representative processes corresponding to gypsum board production, gypsum board paper production, gypsum extraction (i.e. mining), and FGD gypsum production at a coal-fired power plant are presented in this database.

5.4 LCI Related to Material Manufacture

5.4.1 Raw Materials Extraction

Gypsum is mined or quarried in 17 states, of which Oklahoma, Nevada, California, and Indiana account for 62% of the total gypsum mined in the US (Crangle 2014). Approximately 16 MMT of gypsum were mined in 2013 and 90% was used by drywall and plaster product manufacturers (Crangle 2014). Additional uses of mined gypsum include cement production, agricultural applications, and other industrial applications (Crangle 2014). Gypsum is extracted from underground mines, open pits, and quarries of natural gypsum rock. The natural deposit of gypsum rocks are drilled and blasted loose for extraction after removing overlying deposits of soil (Athena 2011). The use of heavy equipment such as front-end loaders, mechanical shovels, and traxcavators and blasting agents require the bulk of the energy demand for gypsum extraction (Venta 1997). Mined gypsum undergoes primary grinding and crushing, typically on-site, into particles of

2 to 5 inches or less (Venta 1997, Athena 2011). Additional processing of the natural gypsum such as screening and drying may be necessary depending on end-use requirements of the material (e.g., cement manufacture, agricultural gypsum).

Table 5-3 presents the inputs and outputs of materials, energy, and emissions from gypsum mining based on data compiled by Athena (2011) from six quarries and one underground mining operation for raw gypsum ore production. Although two of these quarrying operations were located in Canada, a sensitivity analysis conducted by Athena (2011) reported that the source location of natural gypsum had less than a 1% impact on the overall drywall production process LCA. These data, along with additional data (e.g., chemical inputs, fuel and electricity consumption data for processing materials) from the US LCI database and Ecoinvent were used to develop a US gypsum drywall production LCI dataset (Athena 2011). The outbound transport distance of the mined gypsum to the drywall manufacturing plant is not included in Table 5-3. Several sources of international energy and emission data for gypsum mining and processing were also identified (Venta 1997, WRAP 2008, Kellenberger et al. 2004). The process exchange LCI titled "Gypsum Quarry Operation" in the Ecoinvent database presents LCI for gypsum mining and crushing activities for the production of raw material for stucco in Switzerland. Although data for a global context are presented, these data appear to originate from the same source (i.e., Kellenberger et al. 2004). The infrastructure input and mining activity emissions (e.g., particulate matter emissions) are estimated based on information for limestone mining. Waste Resources Action Programme (WRAP) conducted a cradle-to-grave LCA of gypsum plasterboard in the United Kingdom (UK) (WRAP 2008). LCI for imported mined gypsum were obtained from the same source (i.e., Kellenberger et al. 2007) that was used for developing the LCI for Ecoinvent's gypsum quarry operation.

Table 5-3. Athena (2011) – Natural Gypsum Mining LCI (per kg of Mined Gypsum at Mine/Quarry)

Input	Source	Category	Unit	Amount
Explosives	Athena (2011)	Construction and Demolition Debris Management	kg	0.000274
Lubricants	Athena (2011)		kg	1.75E-05
Hydraulic Fluid	Athena (2011)		kg	1.65E-05
Greases	Athena (2011)		kg	3.50E-06
Engine Oil	Athena (2011)		kg	9.50E-06
Antifreeze	Athena (2011)		kg	3.50E-06
Diesel, combusted in industrial equipment	Athena (2011)		L	0.00168
Gasoline, combusted in industrial equipment	Athena (2011)		L	3.33E-05
Propane, combusted in industrial boiler	Athena (2011)		L	2.46E-06
Electricity, at industrial user	Athena (2011)		kWh	0.00133
Natural gas, combusted in industrial boiler	Athena (2011)	Construction and Demolition Debris Management	m^3	0.00182
Fresh Water	Athena (2011)		L	0.15
Recycled Water	Athena (2011)		L	0.142
Transport, truck	Athena (2011)		t*km	0.000316
Transport, Barge	Athena (2011)		t*km	0.0343
Output	**Source**	**Category**	**Unit**	**Amount**

Gypsum, at mine	*Athena (2011)*	Construction and Demolition Debris Management	**kg**	**1**
Overburden (for quarried gypsum)	Athena (2011)		kg	1.94
Other solid rock sold to other industries	Athena (2011)		kg	0.033
Particulates, < 2.5 um	Athena (2011)	Air/Unspecified	kg	9.85E-05
Particulates, > 10 um	Athena (2011)		kg	3.9E-05
Particulates, > 2.5 um, and < 10um	Athena (2011)		kg	6.50E-06
Total suspended solids	Athena (2011)	Water/Stormwater	kg	1.00E-06
Total suspended solids	Athena (2011)	Water/Groundwater	kg	1.15E-09
Oil and Grease , hexane	Athena (2011)		kg	9.00E-10
Chloride	Athena (2011)		kg	5.50E-08
Sulfate	Athena (2011)		kg	2.65E-07
Solid waste	Athena (2011)	Construction and Demolition Debris Management	kg	3.75E-05
Waste oil	Athena (2011)		kg	1.60E-06

5.4.2 Synthetic Gypsum Production

Synthetic gypsum is also used as a raw material in addition to the mined gypsum for drywall production. Currently, mined and synthetic (FGD) gypsum constitute 41% and 57% of the total amount of gypsum used for drywall production in the US, respectively (Athena 2011). The use of synthetic gypsum has increased dramatically in the US from 0.9 MMT in 1993 to 12.3 MMT in 2013 (Balazik 1995, Crangle 2014). Of the synthetic gypsum sold and used in the US, 81% is accounted for in drywall production (Athena 2011, Crangle 2014). Even with the significant increase in the use of synthetic gypsum, approximately 47% of synthetic gypsum produced was landfill in the US in 2013, suggesting significant potential for further growth in recovery and use of synthetic gypsum.

Although a byproduct of many chemical processes such as acid neutralization, citric acid production, sugar production from sugar beets, and titanium dioxide production, synthetic gypsum is predominantly produced from an FGD wet process in coal-fired power plants (Crangle 2013). Calcium bisulfate is produced from the reaction of sulfur dioxide with a calcium-based sorbent used for sulfur dioxide scrubbing in FGD stack. The calcium bisulfate is subsequently oxidized to sulfate dihydrate (i.e., FGD gypsum) by natural or forced oxidation (WRAP 2008). The materials from natural oxidation processes are generally disposed of due to their partial oxidation, while forced oxidation results in a marketable product (WRAP 2008). The oxidation process produces gypsum crystals, which are fractionated using a hydrocyclone to separate larger crystals. The rest of the gypsum crystal suspension is filtered or centrifuged and washed to remove water-soluble substances (Athena 2011). Vacuum filter beds are used to dewater and reduce the moisture content of the FGD to less than 10%.

Although Athena (2011) acknowledges dewatering (to reduce moisture content to less than 10%) as the process that differentiates waste FGD from marketable FGD, WRAP (2008) recognizes oxidation as the process for this distinction. As a result, Athena (2011) attributes inputs for dewatering and transportation of FGD to end-user to synthetic gypsum production, whereas WRAP (2008) assigns inputs and the associated emissions for forced oxidation of calcium bisulfate and the subsequent processing to FGD production. FGD gypsum is not included in the Ecoinvent inventories as only a small quantity of FGD is used in Switzerland (Kellenberger et al. 2007). LCI data reported by Athena (2011) for FGD processing should be used as these are specific to the US. Although Athena (2011) considered FGD dewatering and processing LCI information for conducting an LCA of different drywall products, specific LCI data were not reported. Therefore, because of an absence of any US-specific information, it was not possible to develop an LCI for FGD gypsum production.

5.4.3 Gypsum Paper Manufacturing

Paper layers (facing and backing) sandwich the gypsum core of the drywall. The US-specific and international energy and emission data for gypsum paper production were identified (Athena 2011, Venta 1997, Kellenberger et al. 2007, WRAP 2008). Athena (2011) compiled US-specific energy and materials input for paper production based on data from three gypsum paper plants for 2010. The description of the paper manufacturing processing and the associated LCI presented in this section are based on information and data presented in Athena (2011). Recycled paper sources are exclusively used for manufacturing the paper used for drywall production. Although various paper types (e.g., post-manufacture and post-consumer newspaper, Kraft clippings, mixed waste papers, and old corrugated containers (OCC)) are used for manufacturing facing, backing is exclusively made from OCC.

The process of producing gypsum paper is similar to other paper manufacturing processes. The recycled papers are blended in a pulper to create a slurry of paper fibers. The slurry is cleaned to remove wires, staples, and glue. The pulp is then made into multiple-ply sheets using either rotating cylinders or Fourdrinier flat wire machines. The sheets are pressed together to remove excess water and the residual water is subsequently removed in high-temperature driers. The dried paper is treated with chemicals (e.g., retention chemicals and sizing agents). The treated paper is rolled, trimmed, and packaged. Electricity and natural gas are the major forms of energy used in the production of gypsum paper. Athena (2011) presented materials, energy, and process emission data for 1000 square feet of paper. These values were divided by the weight of 1000 square feet of paper provided by Athena (2011) to estimate data on a mass basis. Table 5-4 presents the materials, energy, and process-related emissions for manufacturing 1 kg of facing and backing paper based on the data compiled by Athena (2011). The outbound transport distance of the finish paper to the drywall manufacturing plant is not included in Table 5-4.

Table 5-4. Materials, Energy, and Process: Related Emissions for 1 kg of Gypsum Paper, at Plant

Input Flow	Category	Unit	Amount	
			(Facing Paper)	(Backing Paper)
Old Corrugated Cardboard (OCC)	Construction and Demolition Debris Management	kg	0.58	1.12
Kraft Clippings		kg	0.143	0
Mixed waste papers/flyleaves, signature/white news blank, magazine blank, coated fly		kg	0.395	0
Starch		kg	9.77E-05	0.000102
Retention chemical (flocculant/coagulant)		kg	0.00196	0.00202
Sizing agents		kg	0.00569	0.00552
Polymer emulsifier		kg	0.000245	0.00026
Other chemicals (defoamers, dyes/fungicide)		kg	0.000299	0.000313
Chemicals used for on-site water treatment (P &N based)		kg	0.000605	0.000583
Packaging materials		kg	0.00238	0.00231
Lubricants		kg	9.16E-05	8.94E-05
Hydraulic fluid		kg	3.51E-06	3.73E-06

Greases		kg	3.63E-06	3.78E-06
Total electricity (purchased and on-site co-generated)		kWh	0.69	0.681
Total natural gas (excluding electricity production)		m^3	0.173	0.176
Diesel fuel oil		L	0.000304	0.000287
Propane		kg	0.000311	0.000323
Fresh well water	Construction and Demolition Debris Management	L	7.71	7.39
Fresh water from "municipality city water system"		L	1.58	1.53
Recycled water re-entering the paper production system		L	2.16	2.26
Water discharged		L	8.95	8.52
Transport, truck		t*km	0.203	0.183
Transport, rail		t*km	0.0199	2.02E-05
Outputs	**Category**	**Unit**	**Amount (Facing Paper)**	**Amount (Backing Paper)**
Gypsum paper, at plant	*Construction and Demolition Debris Management*	*kg*	*1*	*1*
Co-products – downgraded and side rolls	Construction and Demolition Debris Management	kg	0.0295	0.0294
Non-Methane VOCs	Air/Unspecified	kg	8.75E-06	9.16E-06
Total suspended solids	Water	kg	0.000637	0.000619
Biological oxygen demand	Water	kg	0.000794	0.000819
Lead	Water	kg	5.90E-09	6.24E-09
Zinc	Water	kg	1.62E-07	1.72E-07
Copper	Water	kg	6.19E-08	6.58E-08
Total nitrogen	Water	kg	1.24E-06	1.30E-06
Total phosphorus	Water	kg	1.02E-07	1.07E-07
Lead	Soil	kg	1.32E-09	1.38E-09
Non-hazardous solid waste	Waste	kg	0.101	0.101
Hazardous solid waste	Waste	kg	8.73E-06	9.28E-06
Wastewater	Waste	L	5.4	5.18
Sludge waste	Waste	kg	0.0451	0.0424
Solvent mixture waste	Waste	L	6.42E-06	6.82E-06

5.4.4 Gypsum Drywall Manufacturing

The mined and synthetic gypsum is further processed (e.g., secondary crushing, drying, and screening) to produce finely ground gypsum (particle size less than 150 µm) with very low moisture content (Venta

1997). This gypsum stream is calcined (i.e., dehydrated) to produce stucco; calcium sulfate dihydrate transforms to calcium sulfate hemihydrate in the presence of heat (Athena 2011). Stucco may undergo further grinding, after cooling, if necessary. Stucco is mixed with several additives (depending on drywall type), foaming agent, and water to prepare a slurry. The slurry is spread on the paper facing and covered by the paper backing. An automatic knife slices the boards to their desired sizes. The hydrated boards are then transported to a drying kiln to remove the excess water.

While many types of gypsum drywall are produced, as previously described, the two most common products are 1/2" regular and 5/8" Type X gypsum drywall. In 2011, these two types of drywall represented over 80% of all drywall consumed in the US (Crangle 2013). Type X gypsum board is made using noncombustible fibers and is a fire-resistant board (National Gypsum 2013). Athena (2011) compiled materials and energy requirements and process non-energy emissions for 1/2" regular and 5/8" Type X gypsum drywall production from 17 plants in the US. These plants included a mix of small, medium, and large operations as well as a geographical spread of at least one plant in each US census region and represented one-third of the total drywall output in the US in 2010 (by all the 60 plants in the US).

Athena (2011) presented materials, energy, and process emission data for 1000 square feet of 1/2" regular and 5/8" Type X gypsum drywall production. These values were divided by the respective weight of 1,000 square feet of drywall provided by Athena (2011) to estimate data on mass basis. Table 5-5 presents the materials, energy and process related emissions for manufacturing of 1 kg of 1/2" regular and 5/8" Type X gypsum drywall based on the data compiled by Athena (2011). Based on the data presented in Table 5-5, it can be seen that synthetic, mined, and recycled gypsum represented approximately 57%, 41%, and 2% of the gypsum used for drywall production in the US in 2010. Electricity is used at all stages in gypsum board production. Natural gas is primarily used during raw gypsum drying, calcination, and the final product drying (Athena 2011). Particulate matter represents a majority of process non-energy emissions. Athena (2011) also present transport distances of various material inputs and output that can be used to estimate transport-related emissions. Weighted average transport distances for various transport modes were estimated based on the amount and transport distance of various materials provided by Athena (2011). The weighted average one-way distance for each transport mode is also provided in Table 5-5. It should be noted that the outbound transport distance of finished drywall from plant was not included in the cradle-to-gate LCA presented by Athena (2011). The destination of the finish drywall from plant was not provided by Athena (2011). As the gypsum-drywall-manufacturing-related data presented by Athena (2011) are mostly based on surveys of the manufacturers, the outbound distances are probably representative of the average distances to the distribution centers from the plants. The LCI presented in Table 5-5 correspond to the manufacturing and delivery of the drywall to distribution centers.

The materials and energy inputs and emissions data for drywall production process and sub-process (e.g., stucco production) for other countries are available (Venta 1997, Kellenberger et al. 2007, WRAP 2008). The US EPA (2012) used the data reported by Venta (1997) for estimating emission factors for source reduction and recycling of gypsum drywall; Venta (1997) compiled LCI for types of drywalls and associated finishing products based on data collected from drywall manufacturers in Canada.

Table 5-5. Proposed LCI for Gypsum Drywall, at Distribution Center

Input Flow	Category	Unit	Amount	
			1/2" Regular Gypsum Wallboard	5/8" Type X Gypsum Wallboard
Mined/quarried natural gypsum ore (US source)	Construction and Demolition Debris Management	kg	0.265	0.263
Mined/quarried natural gypsum ore (imported from Canada/Mexico)		kg	0.119	0.123
Synthetic gypsum (FGD)		kg	0.544	0.549
Post-consumer gypsum		kg	0.0177	0.0186
Transport, barge		t*km	0.381	0.393
Transport, truck		t*km	0.496	0.493
Transport, rail		t*km	0.216	0.218
Transport, conveyor		t*km	0.00128	0.00127
Starch	Construction and Demolition Debris Management	kg	0.00459	0.00324
Vermiculite		kg	0.00495	0.00438
Fiberglass		kg	0.000447	0.00246
Dispersant		kg	0.00311	0.00244
Retarder	Construction and Demolition Debris Management	kg	0.000576	0.000368
Potassium Sulfate		kg	0.000218	0.000115
Dextrose		kg	0.000724	0.000407
Clay, kaolin		kg	0	0.000572
Boric Acid		kg	0.00014	5.86E-05
Land Plaster		kg	0.00151	0.000689
Foaming agent (soap)		kg	0.000791	0.000595
BM Accelerator		kg	0.00107	0.000622
Ammonium Sulfate		kg	7.02E-05	4.50E-06
Edge Paste		kg	0.000402	0.000288
STMP		kg	7.66E-05	5.41E-05
Shredded Paper		kg	0.000357	0.000189
Talc		kg	2.49E-07	1.76E-07
Paper End tape		kg	0.000451	0.000449
Ink (water based)		kg	3.10E-06	3.09E-06
Ink (oil based)		kg	1.98E-07	1.95E-07
Ink (alcohol based)		kg	2.32E-06	2.32E-06
Shrink-wrap		kg	2.87E-05	2.86E-05
Plastic slip sheets		kg	2.78E-05	2.82E-05
Rail bags		kg	4.16E-05	4.10E-05
Other Plastics		kg	1.72E-05	1.74E-05
Cardboard Edge Protectors		kg	1.37E-06	1.37E-06
Plastic Banding		kg	5.92E-07	6.04E-07
Steel Banding		kg	1.44E-06	1.47E-06
Zip tape		kg	1.05E-05	1.05E-05
Dunnage/Bunks/Sleutters		kg	0.0161	0.016
Adhesive for Dunnage/Bunks/Sleutters		kg	2.63E-07	2.65E-07
Motor Oils		kg	1.23E-06	1.18E-06
Gear Oil (Transmission)		kg	1.66E-06	1.61E-06
Lubricants		kg	3.57E-06	3.56E-06
Hydraulic Fluid		kg	4.66E-08	4.64E-08
Greases		kg	1.38E-07	1.26E-07
Antifreeze		kg	8.10E-07	8.11E-07

Input Flow	Category	Unit	Amount	
			1/2" Regular Gypsum Wallboard	**5/8" Type X Gypsum Wallboard**
Locomotive Oil	Construction and Demolition Debris Management	kg	1.71E-06	1.56E-06
Electricity		kWh	0.0661	0.0663
Natural gas		m3	0.0723	0.0723
Diesel fuel oil		L	0.000233	0.000236
Propane		kg	0.00617	0.00623
Gasoline		L	0.0514	0.0519
Fresh water (process)	Construction and Demolition Debris Management	kg	0.576	0.566
Reclaimed water (process)		kg	0.0252	0.0248
Fresh water used for cooling or steam production		L	0.537	0.533
Output Flow	**Category**	**Unit**	**Amount**	**Amount**
Gypsum Drywall	*Construction and Demolition Debris Management*	*kg*	*1*	*1*
Internal gypsum waste -recycled back into the production system	Waste	kg	0.027	0.0273
Off-spec GWB used as BDS	Construction and Demolition Debris Management	kg	0.00977	0.00977
Total Particulate Matter (PM)	Air/Unspecified	kg	6.45E-05	6.49E-05
as PM10	Air/Unspecified	kg	4.67E-05	4.73E-05
as PM2.5	Air/Unspecified	kg	1.78E-05	1.77E-05
VOC	Air/Unspecified	kg	3.25E-06	3.39E-06
Lead	Air/Unspecified	kg	1.77E-08	1.76E-08
Mercury	Air/Unspecified	kg	9.70E-09	9.91E-09
Total suspended solids	Water/Unspecified	kg	2.39E-08	2.26E-08
Total Organic Carbon	Water/Unspecified	kg	1.06E-08	1.09E-08
Lead	Water/Unspecified	kg	2.61E-14	2.68E-14
Zinc	Water/Unspecified	kg	1.24E-11	1.12E-11
Copper	Water/Unspecified	kg	2.07E-12	1.87E-12
Sulfates	Water/Unspecified	kg	3.89E-09	3.51E-09
Sulfide	Water/Unspecified	kg	1.19E-10	1.08E-10
Oil & Grease	Water/Unspecified	kg	8.49E-09	8.33E-09
Ammonia	Water/Unspecified	kg	1.31E-09	1.34E-09
Wastewater to waste treatment facility	Construction and Demolition Debris Management	L	9.30E-05	9.15E-05
Solvent mixture waste to incinerator		L	2.94E-05	2.95E-05
Sludge waste to landfill		kg	1.42E-05	1.44E-05
Non-hazardous solid waste (including packaging) to landfill		kg	0.00174	0.00173
Other(s) solid waste		kg	0.00185	0.00182
Paper to recycler		kg	0.00019	0.000194
Plastic to recycler		kg	3.62E-06	3.67E-06
Wood to recycler		kg	8.93E-05	9.10E-05
Steel scrap to recycler		kg	3.41E-05	3.42E-05
Hazardous solid waste to incinerator		kg	1.70E-06	1.66E-06

5.5 LCI Related to Disposal

The emissions from gypsum drywall landfill disposal result from operating landfill equipment during material and cover soil compaction and placement, including both fuel-related and pre-combustion emissions, as well as those associated with the physical, chemical, and biological decomposition of gypsum drywall in landfill. The exposure to precipitation or other liquids (e.g., landfill leachate) is expected to result in leaching emissions. The liquids and gaseous emissions to the environment are expected to depend on the biogeochemical environment of the landfill (e.g., MSW landfill, CDD landfill), as well as in-place environmental controls employed at the landfill.

WARM estimates landfilling emission factors related to drywall, which include GHG emissions from transportation and the operation of the landfill, carbon sequestration, and methane generation from biological decomposition of the facing and backing paper. WARM does not consider liquid emissions from landfills. In addition to emissions from heavy equipment, US EPA (2012) included fugitive methane emission associated with drywall paper decomposition from drywall disposed of in a CDD landfill site without GCCS. US EPA (2012) used methane generation potential reported by Staley and Barlaz (2009) to assess methane generation from drywall. Staley and Barlaz (2009) estimate the methane generation potential of drywall by multiplying the methane generation potential of OCC/Kraft bag reported by Eleazer et al. (1997) by 0.1 to account for the relative mass of paper (approximately 10% of drywall by mass based on the data reported by National Gypsum Company (2008)) and adjusted for the paper-specific moisture content (6% by mass reported by Tchobanoglous et al. (1993)); the drywall methane generation potential was estimated to be 15.2 m^3 per dry MT of drywall. The US EPA (2012) assumed a methane oxidation rate of 10% in the landfill cover based on work by Czepiel et al. (1996) to estimate fugitive emissions from methane generation. Additional details on the information used in US EPA (2012) to estimate methane and carbon dioxide emissions from the disposal of drywall in a CDD and MSW landfill are presented in Section 2.5.10.8. The CDD landfill methane and carbon dioxide emissions are estimated as 0.010 and 0.034 kg, respectively.

The US EPA (2012) considered only methane emissions from drywall decomposition in anaerobic landfill environments for estimating GHG impacts. The production of hydrogen sulfide from the biological decomposition of organic matter in anaerobic conditions in the presence of dissolved sulfate (primarily from gypsum) has been reported to be a major environmental concern associated with gypsum drywall disposal in landfills (Jang 2000, Xu 2005). Several factors, including moisture content, organic content, pH, and temperature, may contribute to the production of hydrogen sulfide in landfills (Elsgaard et al. 1994, Knoblauch and Jorgensen 1999, Koschorreck 2008). Several studies have indicated that the amount of organic matter present in CDD landfills, although significantly lower than in MSW landfills, is not a limiting factor for hydrogen sulfide production; the paper backing on drywall is sufficient to sustain a viable microbial community that produces hydrogen sulfide (Hardy Associates 1978, Townsend 2002, New Hampshire Department of Environmental Services 2004). Tolaymat et al. (2013) reported a decay rate constant for drywall decomposition and the associated hydrogen sulfide production. Xu (2005) assessed the impact of different cover materials in reducing hydrogen sulfide emissions from CDD landfills and reported the hydrogen sulfide concentration in gaseous emissions from drywall decomposition based on laboratory experiments.

Anderson et al. (2010) evaluated hydrogen sulfide emission and sulfur content of CDD fines from nine landfills in the US to estimate the potential for and rate of hydrogen sulfide generation from disposal of CDD fines, based on sulfur content. Anderson et al. (2010) emission of 5,360 ft^3 of H_2S per ton of sulfur disposed of in landfill. From stoichiometry, sulfur represents 18.6% of the gypsum (by weight) and assuming that drywall is comprised of 92% gypsum (Marvin 2000), this approximately equates to 0.041 kg of hydrogen sulfide release per kg of drywall disposal (using a density of about 1.42 grams per liter of hydrogen sulfide at 20^{o} C and 1 atm pressure). Plaza et al. (2007) estimated the attenuation of hydrogen sulfide by different landfill cover materials, included clayey and sandy soils; the average attenuation of these two soils was 47.5%. Additional details on the studies conducted by Anderson et al. (2010) and Plaza et al. (2007) are provided in the RSM chapter, Section 9.5. Using the same assumptions listed in Chapter 9, and in the absence of other US-data, the average estimated hydrogen sulfide release rate based on these studies (including the estimated average cover soil removal efficiency) is 0.021 kg hydrogen sulfide per kilogram of drywall disposed of in a CDD landfill.

Ecoinvent provides LCI which are gypsum specific for disposal in inert material landfills and sanitary (MSW) landfills. Although the database includes LCI for Switzerland as well as the global case, the LCI sets appear to be based on the data published by Doka (2009) based on the management practices in Switzerland. Doka (2009) presented LCI for three EOL management options for building materials. In the first option, the building materials are source segregated and recycled; no disposal is assumed to occur in this option. In the second option, the building materials are transported and processed at a material recovery facility and the materials that cannot be recycled are disposed of in a landfill. The fine fractions recovered in this option are assumed to be disposed of at a sanitary landfill; the gypsum content of the recovered fines appears to be primary reason for the requirement of fines disposal in a sanitary landfill. In the third option, building materials are assumed to be disposed of in an inert debris landfill without resource recovery. Liquid emissions are considered only for sanitary landfills and not for inert debris landfills. For disposal in sanitary landfills, Doka (2003) estimated that although 100% of gypsum will decompose, 56.2% of the dissolved sulfate reduces and precipitates as sulfides and eventually 6.5% of the sulfur is emitted to the air. Although not specifically mentioned by Doka (2009), it seems that the balance of sulfur (37.3%) is assumed to exit the landfill with leachate. Doka (2009) does not provide emission factors for hydrogen sulfide.

The GaBi database contains processes for three separate landfill exchanges (three separate LCI datasets) for CDD material disposal. These are not specific to the drywall product and the processes are based on data from countries within the EU-27 region. WRAP (2008) uses the emissions data presented by Golder Associates (2007) for gypsum drywall disposal in landfill. Golder Associates (2007) compiled liquids and gaseous emission from gypsum drywall disposal in monofill as well as co-disposal with MSW in UK based on LandSim and GasSim2 modeling results, respectively. Leachate is assumed to be treated before discharge into the environment. The emissions from landfill construction, operation, and closure were based on those used by WRATE. No interaction between the MSW and drywall was assumed for leachate and landfill construction and operation were assumed. The impact of MSW on gas generation from drywall in co-disposal scenario was modeled; hydrogen sulfide emissions from drywall co-disposed of with MSW was estimated to be more than four times that from a monofill. A time horizon of 150 years with active gas collection for the MSW landfill model was assumed. Based on GasSim modeling, Golder Associates (2007) reported emission estimates of 0.5 g and 2.4 g of hydrogen sulfide per MT of plasterboard disposed of in a monofill and co-disposal, respectively.

Jang (2000) conducted batch and column leaching tests of several individual CDD materials, including new gypsum drywall, using SPLP extraction fluid (and US EPA SW-846 Method 1312). Drywall material was cut into square pieces approximately 5 cm on each side. Results from these experiments were used to estimate liquids emissions that would result from disposing of gypsum drywall in an unlined landfill. Unlined landfill disposal may occur due to the classification of CDD type wastes as more chemically inert than MSW components and in some areas the term *inert material landfill* is used for CDD sites (Doka 2003). Batch test data were used for the constituent with concentrations measured above detection limits due to the greater L:S ratio in batch tests (20:1 vs. 5.3:1 for column tests); higher L:S ratios are designed for and are generally capable of leaching a higher quantity of the total constituent mass from the solid material (i.e., drywall). Batch test concentrations were multiplied by the total solution volume and divided by the sample mass to estimate leachability on a per-kilogram-drywall basis (Table 5-6).

Column leaching test data were used to estimate the leaching emission of potassium and magnesium, as these were not detected in batch leaching tests. The column leaching test entailed percolation of approximately 160 L of SPLP extraction fluid through a 30-cm-diameter polyvinyl chloride column containing 30 kg of drywall over a 3-month period; an L:S ratio of 5.3 was achieved in this test. The amount of potassium and magnesium leached between two sampling events was estimated by multiplying the leachate volume collected since the previous sampling event with the measured concentration. The cumulative leaching amount was estimated by adding the leaching amount for each sampling intervals.

Batch test results for nitrite and nitrate solution concentrations were not included due to the presence of nitric acid in the SPLP leaching solution. Although sulfate is present in the SPLP leaching fluid, the sulfate concentrations in drywall leachate from batch tests were approximately one to three orders of magnitude greater than those observed for all other CDD materials (e.g., levels of 1.9, 125, and 1,430 mg/L were found in leaching fluid for aluminum, insulation, and drywall, respectively). Therefore the sulfate contributed by the leaching fluid was considered insignificant compared to the amount leached from drywall. High levels of calcium in comparison to other CDD materials are likely attributable to the dissolution of calcium and sulfate present in drywall; the mass ratio of SO_4/Ca found in the SPLP fluid after batch tests (approximately 2.6) approximately matched that of the SO_4/Ca mass ratio in raw gypsum molecules of 2.4 ($CaSO_4 \cdot H_2O$).

The gypsum drywall leaching data and energy consumption data from landfill operations were used to develop an LCI process dataset for disposal of drywall at an unlined CDD landfill, as presented in Table 5-6. Emissions are provided per kilogram "Gypsum drywall, at unlined CDD landfill" flow. Although the actual emissions are expected to be greater than the estimated liquids emission as the material would be subjected to leaching a higher L:S ratio than used by Jang (2000) for the batch leaching test, using these emissions for LCA until the total emission estimates become available would be more accurate than excluding liquids emission altogether. The methane emission rate estimate presented earlier in the section is included in Table 5-6. Details on the diesel and electricity consumption included in the "CDD landfill operations flow" and on how the cover soil requirement was determined are provided in Chapter 2. The density of bulk gypsum drywall for the cover soil requirement estimation was provided by CCG (2006). In the absence of average nationwide distance data, the site of gypsum drywall removal was assumed to be located 20 km from the CDD landfill disposal site.

Table 5-6. LCI Dataset: Gypsum Drywall Disposal, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
Gypsum drywall, from building removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.0686
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Gypsum drywall, at unlined CDD landfill			*kg*	*1*
Methane	Staley and Barlaz (2009)	Air/Unspecified	kg	0.010
Carbon dioxide	US EPA (2012)	Air/Unspecified	kg	0.034
Hydrogen sulfide	Anderson et al. (2010)	Air/Unspecified	kg	0.021

COD	Jang (2000)	Water/Groundwater	mg	1,160
Chloride	Jang (2000)	Water/Groundwater	mg	152
Sulfate	Jang (2000)	Water/Groundwater	mg	2.86E4
Sodium	Jang (2000)	Water/Groundwater	mg	214
Potassium	Jang (2000)	Water/Groundwater	mg	21
Magnesium	Jang (2000)	Water/Groundwater	mg	66
Calcium	Jang (2000)	Water/Groundwater	mg	1.1E4

5.6 LCI Related to Recycling

Drywall recycling has been gaining momentum as landfills place more restrictions on drywall disposal due to actual or potential odor issues. Drywall scrap can be processed and reused in a variety of applications, such as a soil conditioner and liming agent and production of new drywall. Discarded drywall sources include off-spec drywall generated at the gypsum board plant as well as scraps from new construction or renovation or material produced from structural demolition. However, because of the quality issues described previously, the majority of waste gypsum used for recycling comes from the plant off-spec material and scraps from construction and renovation projects. At drywall manufacturing plants, scrap drywall generated from off-spec boards is recycled back into the manufacturing process for new drywall material. The drywall generated from demolition activities is not recycled as commonly as drywall from other sources due to possible contamination from other CDD materials such as nails, paint, and joint compound (Venta 1997, Cochran 2006). For developing emission factors for drywall recycling for the WARM model, the US EPA (2012) assumed that 19% and 81% of the recycled drywall is recycled for new drywall manufacturing and soil amendment production, respectively. However, the US EPA (2012) only considered recycling of new drywall scraps generated from construction sites.

Drywall discarded during installation of the interior wall in new construction or renovation can be more readily recycled due to ease of separation from other CDD materials. Processing consists of size reduction, where initial size reduction may occur during material handling with heavy equipment while further size reduction occurs through crushing and grinding. Based on a review of the operation of a few drywall processing facilities and trial of various methods, Townsend et al. (2001) reported that the rotating action of a trommel screen is sufficient to separate paper and pulverize the gypsum core. A material recovery rate of approximately 70% was reported for this processing method based on trial operation in Florida. As paper constitutes approximately 5.5% (by weight) of new drywall, the rest (30%) of the residue primarily consists of unrecovered gypsum. The separated paper can be recovered for the production of new paper for drywall manufacturing, and the recovered gypsum is directly used as raw material for new drywall manufacturing (Venta 1997, WRAP 2008).

Energy requirements for various equipment used for drywall processing have been presented by Cochran (2006). However, the throughput rate for this equipment was not provided by Cochran (2006) to estimate the energy requirement per unit weight of drywall processed. WRAP (2008) presented energy requirement and recovery rates for processing post-manufacture and post-consumer drywall based on data provided by three plasterboard manufacturers and four recyclers in the UK. The electricity and diesel demand of 9.9 kWh and 0.9 L per MT of post-consumer drywall processing was reported, respectively. The electricity and diesel requirement for post-manufacture drywall processing was reported to be 9.6 kWh and 1.3 L per MT drywall, respectively. The processing of 1 MT of post-consumer drywall was reported to yield 930 kg, 68 kg, and 2 kg of recycled gypsum, reclaimed paper, and waste. Details on the process used are not provided by WRAP (2008). The US EPA (2012) used the post-consumer drywall processing electricity and diesel requirement and recovery rate for developing emission factors for drywall recycling for WARM.

The emission associated with the transport of gypsum and paper recovered from processing gypsum drywall should be considered for LCA. The US EPA (2012) reported that it used the transport distance of the finished drywall transport compiled provided by the US Census Bureau (2004) as a proxy for the transport distance of discarded drywall to a recycling facility and recycled gypsum to a drywall manufacturing facility. However, the commodity flow survey does not provide transport distances specific to drywall. The commodity closest to drywall that this survey provides distances for is "lumber and other construction materials."

Doka (2009) (Ecoinvent) presented LCI for three EOL management options for building materials. In the first option, the building materials are source segregated and recycled; no disposal is assumed to occur in this option. In the second option, the building materials are transported and processed at a material recovery facility and the materials that cannot be recycled are disposed of in a landfill. In the third option, building materials are assumed to be disposed of in an inert debris landfill without resource recovery. These processes include burdens associated with the dismantling process (e.g., energy used and particulate matter from the process). The second option includes burdens associated with waste sorting. None of these options includes the burden associated with processing recovered building materials to produce a recycled product. For example, burdens associated with the processing of drywall to produce recycled gypsum and paper are not included in these LCI.

In addition to use in new drywall manufacturing, recycled gypsum could be land-applied in agricultural applications or used for cement production. The liquids emissions presented in Table 5-6 can be used as a proxy for the liquid emissions from land application of recycled gypsum. As recycled gypsum offsets production and use of natural gypsum in these applications, emissions from natural mined gypsum should be considered as well for LCA. Isaac and Morris (2012) conducted Leaching Environmental Assessment Framework (LEAF) leaching tests on four mined gypsum samples to assess metals leaching associated with the use of recycled gypsum as soil amendment. Energy and materials inputs and particulate matter emission associated with land application of recycled gypsum are lacking; the "Diesel, combusted in industrial equipment" input flow is included in the dataset as a placeholder until this energy input can be quantified. Table 5-7 presents liquid emissions LCI associated with land application of recycled gypsum. It is assumed that the recycled gypsum is transported 20 km from processing facility for agricultural application.

Table 5-7. LCI Dataset: Recycled Gypsum Land Application as Agricultural Amendment

Input Flow	Source	Category	Unit	Amount
Gypsum Drywall		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Diesel, combusted in industrial equipment		Flows	L	0
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Recycled gypsum- land applied as agricultural amendment			*kg*	*1*
COD	Jang (2000)	Water/Groundwater	mg	1,160
Chloride	Jang (2000)	Water/Groundwater	mg	152
Sulfate	Jang (2000)	Water/Groundwater	mg	2.86E4
Sodium	Jang (2000)	Water/Groundwater	mg	214
Potassium	Jang (2000)	Water/Groundwater	mg	21
Magnesium	Jang (2000)	Water/Groundwater	mg	66
Calcium	Jang (2000)	Water/Groundwater	mg	1.1E4

5.7 Data Gap analysis and Opportunities for Additional LCI Data

Table 5-8 summarizes the type of data presented by various sources reviewed for compilation of drywall EOL management LCI. Only Cochran (2006), Athena (2011), and the US EPA (2012) provide information with respect to US-based processes. Some sources used data from the other sources presented in. For example, the US EPA (2012) used data from Venta (1997) and WRAP (2008). As shown in the table, many sources present only part of the data/information needed for LCI compilation. For example, WARM presents only GHG emissions and uses emissions only associated with fuel consumption in equipment to estimate the landfill emission factor. Similarly, Ecoinvent only has partial landfill leachate emissions data because leachate from inert materials landfills are not considered.

A majority of LCI information available on drywall pertains to the manufacturing aspects of the life cycle. Only limited EOL-specific LCI are available. Based on a review of the available information, the following data gaps were identified for compiling a more comprehensive LCI dataset for drywall EOL management:

1. **Long-term leachable emissions from drywall placed in a landfill.** As described earlier, the liquid emissions presented in this study are based on SPLP tests, which simulate leaching from land-application or disposal in an inert debris landfill. The batch leaching data used for estimating liquid emissions correspond to L:S ratio of 20 and, therefore, do not represent complete liquid emission. As gypsum drywall is typically disposed of with other discarded materials and not disposed of in a monofill, field-scale leachate quality data specific to gypsum drywall disposal in landfill are not available and probably will not be available in the future. The liquid emissions from gypsum drywall placement in an inert materials as well as MSW landfills would, therefore, need to be based on laboratory-scale studies simulating long-term liquids emissions. Published leaching studies have been conducted on gypsum drywall as a component in the CDD debris waste stream (Jang 2000, Jang and Townsend 2003).

2. **Materials and energy input and emission from gypsum drywall processing.** Only one source (Cochran 2006) identified energy requirement (MJ/hour) of drywall processing equipment based on a survey of a handful of equipment manufacturers. Due to the lack of throughput data (MT/hour), energy requirement per unit mass of drywall could not be estimated. Some of the data (e.g., consumables, fuel and electricity usage, water consumption, material throughput) tracked by the facility owner from financial accounting perspective can be readily used for developing more comprehensive LCI for drywall processing.

   The process non-energy emission (e.g., particulate matter emission from drywall grinding, water consumption for dust control) associated with drywall grinding are lacking. Future research should focus on collecting and compiling these data.

3. **Long-term gaseous emissions from drywall placement in landfills**. The disposal of gypsum drywall in inert debris and MSW landfills is expected to produce methane and hydrogen sulfide. Staley and Barlaz (2009) used the methane generation potential of OCC/Kraft paper as a proxy for the methane generation potential of gypsum paper. The US EPA (2012) used the methane generation potential estimate provided by Staley and Barlaz (2009). Although some sources presented the gypsum drywall decay rate, the hydrogen sulfide generation estimates are lacking.

Table 5-8. Overview of LCI Data Available

Process	**Venta (1997)**	**Athena (2011)**	**US EPA (2012) WARM**	**Cochran (2006)**
Gypsum Mining	X	X	P	
Paper Backing Production	X	X		
Drywall Manufacturing	X	X	X	
Landfilling			P	
Drywall Grinding and Paper Screening			X	P
Paper Recovery and Recycling				
Transportation		X	X	

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6 Wood

6.1 Introduction

Wood is the third most widely used construction material in the US after asphalt concrete and PCC (Cochran and Townsend 2010). Softwood and hardwood lumber constituted approximately 58% (by weight) of 60 MMT of the solid wood products manufactured in the US in 2011 (Howard and Westby 2013). Approximately 46% and 14% of the total solid wood products consumed in the US in 2009 were used for residential (single family, multifamily, mobile home) and non-residential building construction and renovation, respectively (McKeever and Howard 2011). Light wood framing utilizing dimensional lumber and engineered wood is employed heavily in residential construction (Wacker 2010, McKeever and Howard 2011). This section covers the following solid wood product wastes, which commonly appear in CDD, including dimensional lumber and engineered wood products:

- plywood and oriented strandboard (OSB),
- particleboard,
- medium-density fiberboard (MDF),
- structural laminated veneer lumber,
- glue laminated timber
- wood I-joists

Woody wastes from land clearing debris (LCD) activities are covered in Chapter 7 of this report. Paper products, although representing the largest wood-derived product stream, represent a very small fraction of discarded CDD materials by weight and, therefore, are not included in the scope of CDD materials investigated in this report. A large amount of wood waste (wood residues) is also generated during manufacturing and more than 98% of these are used by the wood products manufacturing industry as fuel or as a feedstock for other products such as particle board. In an LCA context, the management of these residues and the associated environmental impacts are typically attributed to the product manufacturing phase (Puettmann and Wilson 2005). The LCI associated with the management of this portion of the wood waste stream are not presented in this report.

Figure 6-1 depicts the processes in the life cycle of wood products, beginning with raw materials production and extraction from forestry and silviculture (active management of forest resources) operations and harvesting, through processing into product, use, and EOL management. The production phase consists of multiple sub-processes. The nature and complexity of sub-processes depend on the type of wood used and the desired end product.

The wood products used for construction are discarded at the end of their service life, which is reported to range from 50 to 100 years (Cochran and Townsend 2010). As depicted in Figure 6-1, the EOL management options of discarded wood products include landfill disposal (either in MSW or dedicated CDD landfills), combustion, recycling, and composting. Although not a common practice, the discarded wood products can be used for the production of new wood products (MGE 1997, Cochran 2006, US EPA 2012).

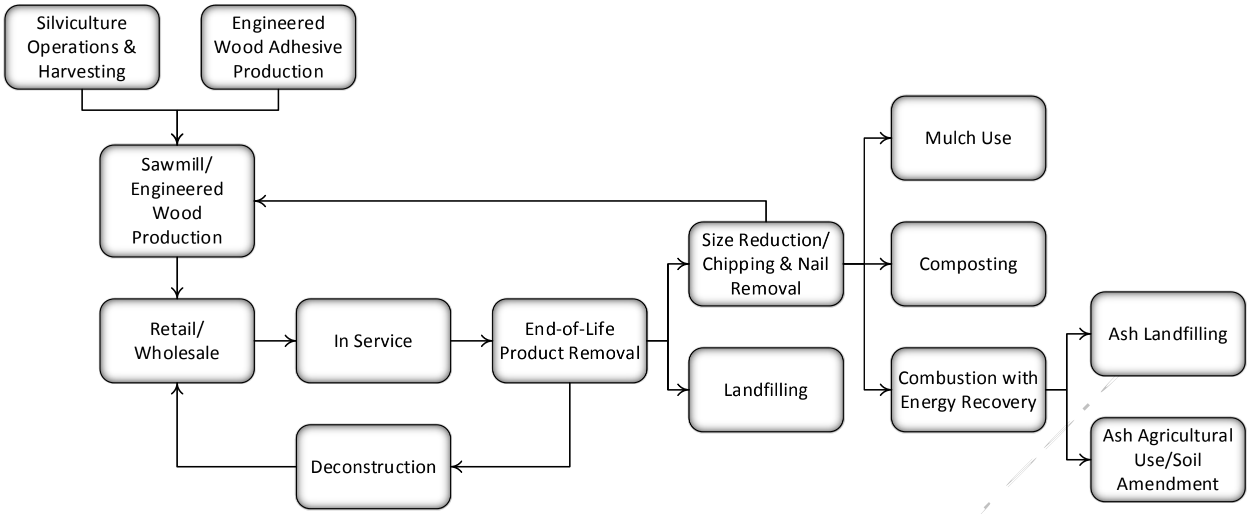

Figure 6-1. Life-Cycle Material Flows and Processes for Wood Product Manufacturing and EOL Management

6.2 EOL Management

Wood, along with other construction materials, is discarded during construction, renovation, demolition and/or deconstruction of the built environment. Based on a review of the data from 11 state-level waste composition studies, Staley and Barlaz (2009) reported that lumber comprised approximately 40% of discarded CDD by mass. Several estimates of discarded CDD wood materials have been published (Wiltsee 1998, Cochran an Townsend 2010, Falk and McKeever 2012). Wiltsee (1998) conducted telephone surveys of demolition and land clearing contractors in 30 metropolitan areas in the US with populations ranging from approximately 100,000 to 3.9 million to estimate the wood generation rate and management options (including CDD wood waste); CDD wood was defined to include LCD. Wiltsee (1998) estimated the weighted-average CDD wood waste generation rate to be 0.069 MT per capita per year, which is equivalent to an annual generation rate of approximately 22 MMT for the US.

Falk and McKeever (2012) reported that CDD debris generated in 2010 included approximately 33 MMT of wood; approximately 80% of the CDD wood was attributed to demolition activities. These estimates were based on economic metrics (e.g., population change, housing completions) and construction activities and unit generation rates (debris per capita per year). Cochran and Townsend (2010) estimated generation of 36 to 55 MMT of CDD wood waste in 2002 based on a materials-flow analysis approach. Based on a compilation of disposal and recycling data from individual states and regional-scale composition studies of CDD debris landfilled, US EPA (2014) estimated that approximately 24 MMT of CDD wood was managed by disposal and CDD processing facilities in the US in 2011; approximately 74% and 26% of the discarded CDD wood (excluding LCD) was landfilled and recycled (including combustion), respectively. Several other studies have published data pertaining to EOL management of discarded CDD materials. Wiltsee (1998) estimated that approximately 70%, 15%, and 11% of the CDD waste wood was landfilled or incinerated, mulched, and used as biomass fuel, respectively; the fraction of wood incinerated with energy recovery was not provided. Approximately 4% of the CDD wood waste was estimated to be used for production of pulp chips, color mulch, pressed fire logs, and fuel pallets.

The type of modification to the built environment (e.g., construction, renovation, or demolition) impacts the relative fraction of wood waste present in the waste stream, which in turn impacts the viable EOL management processes. The fraction of wood in CDD waste generated from construction, renovation, and demolition of residential structures has been reported to be higher than for nonresidential structures

(Cochran et al. 2007). Cochran et al. (2007) also reported a greater wood fraction in the waste stream from renovation than from demolition of residential and non-residential buildings.

The material handling at the point of generation (e.g., segregation from other CDD materials) impacts the quality and in turn EOL management options. For example, the recovery and recycling of wood commingled with other CDD materials may not be economically viable because of the extensive processing that would be required. In addition to contamination by other materials, the presence of treated wood may also dictate the EOL management options.

Landfilling of wood wastes appears to be the most common EOL management strategy employed in the US. Combustion of discarded CDD wood seems to be practiced on a limited scale and is often considered a form of recycling (MGE 1997, Falk and McKeever 2004, Cochran 2006). Creosote, used extensively for treating wood used for railroad ties, has been reported to increase the wood energy content (Smith and Bolin 2010). The presence of treated wood affects all EOL management options. For example, ash from the combustion of wood that includes treated wood may be limited in viability as a soil amendment (Solo-Gabriele and Townsend 1999).

Discarded CDD wood is not commonly reused in the US. Building deconstruction as an alternative to demolition has been proposed and practiced on very small scale to enhance material recovery and reuse (NAHB 1997, Denhart 2010). Closed loop recycling of wood, unlike many other waste materials, is severely limited (MGE 1997). Although engineered wood products can be manufactured using discarded dimensional lumber, dimensional wood cannot be manufactured using engineered wood products due to the processes wood undergoes when engineered wood products are manufactured. Recycling dimensional lumber may also involve processing to smaller pieces of lumber or size reduction (i.e., chipping) to produce engineered wood products (Merrild and Christensen 2009). This downcycling has been referred to as *a wood cascade chain* (Sathre and Gustavsson 2006, Hoglmeier et al. 2013) and entails energy and carbon balances considering land use, primary material substitution, transit, and manufacturing considerations.

Open-loop recycling generally involves size reduction for the production of mulch or chips. Mulch can be used in erosion control, as a bulking agent in composting, as a boiler fuel, or for decorative purposes and to assist in soil moisture retention in landscaping and gardening. Composting, although used for other woody wastes (i.e., LCD discussed in Chapter 7), is rarely practiced for CDD wood materials (US EPA 2012). Table 6-1 presents processes that should be considered for LCA of EOL management options for wood products.

Table 6-1. LCI Needed for LCA of Wood Products EOL Management

Process	Description
Wood Products Manufacturing	Major operations of wood products manufacturing include forestry operations, timber harvesting and processing timber at a mill. A variety of wood products are manufactured in the US; cradle-to-gate materials and energy inputs and emissions depend on the type of wood processed and the type of end product.
Transport	The fuel requirements and emissions associated with the transport of CDD wood from the point of generation to a recycling facility or a landfill, and from a recycling/processing facility to end users (e.g., for the use of mulch or wood chips for combustion) should be considered for LCA of wood products.
Landfilling	The materials (e.g., equipment, soil, water) and energy (fuel, electricity) inputs for placement and compaction of CDD wood in a landfill along with process non-energy emissions (e.g., particulate emissions from equipment operation, gas emissions from the decomposition of wood, and liquids emissions

	associated with biogeochemical degradation of CDD wood in a landfill) should be included in LCA. Energy recovery from the collection and combustion of landfill gas should also be accounted for.
Recycling	CDD wood is typically recycled into either mulch or biomass fuel; materials (e.g., equipment), fuel consumption, and fuel and non-fuel emissions from processing CDD wood in a chipper or grinder and storing the processed materials should be considered. Land application of mulch, similarly to landfilling of CDD wood, is expected to generate leachate influenced by precipitation.
Combustion	Fuel and non-fuel emissions from combusting biomass fuel produced from CDD wood include those generated during the drying of biomass fuel and blending with other woody materials prior to combustion, and from combustion.
Ash Management	Wood ash produced from the combustion of CDD wood in a boiler is typically managed by landfilling, although land application and other uses may occur. Materials and emissions from management of the ash landfill will include similar considerations as landfilling CDD wood, however gaseous emissions are not expected.

6.3 LCI Sources

Peer-reviewed literature, government and private industry publications, and various LCA modeling tools were reviewed to identify available LCI datasets pertaining to CDD wood EOL management. Table 6-2 lists data sources reviewed to compile LCI presented in this chapter. If LCI data were not available, process metadata and documents were reviewed to evaluate the completeness of the dataset (e.g., emissions categories included, background data used to compile the dataset, geographic location and time period of the data). The primary sources of information used to develop the LCI datasets and information identified, if available, were reviewed. Although LCI from many information sources listed in Table 6-2 may not pertain specifically to the US, these sources are presented and discussed for better understanding of the inputs used to develop these LCI and the LCI information available globally.

Table 6-2. LCI Sources to Develop Wood Products LCI

LCI Source	Description
AP-42 (US EPA 1995a)	US EPA (1995a) provides air emissions factors for plywood manufacturing, reconstituted wood products (OSB, particleboard, medium density fiberboard or MDF, hardboard and fiberboard) manufacturing, wood preservation, and manufacture of engineered wood (including glulam, laminated veneer lumber, and others).
US LCI (2012)	The US LCI (2012) database contains US-specific LCI data for solid wood products related to extracting raw materials, logging, and manufacturing wood products. The wood LCI primarily originate from the Consortium for Research on Renewable Industrial Materials (CORRIM) publications. CORRIM's work included LCI development for different wood products in different geographic areas within the US, reflected in the unique inputs/outputs for each process.
Cochran (2006)	Cochran (2006) compiled energy requirements for recycling waste wood into mulch and incineration as boiler fuel, identified as the primary EOL management option employed in the US.
Dubey et al. (2010)	Dubey et al. (2010) presents leaching data for four types of pressure-treated wood simulating contaminant leaching under a variety of recycling and disposal scenarios.
GaBi	GaBi presents wood product manufacture related datasets that are primarily the same as those in the US LCI (2012) database. Wood-specific elemental data for untreated and treated wood are provided on a mass basis within the GaBi process datasets related to wood product

	landfilling.
Hasan et al. (2010)	Hasan et al. (2010) present arsenic, copper, and chromium leaching data for untreated, weathered chromated copper arsenate (CCA)-treated, and ACQ-treated wood subjected to natural precipitation.
Jambeck (2004)	Batch tests conducted on CCA-treated wood at different L:S ratios with deionized water as the leaching fluid were undertaken and metals leached were quantified.
Jang (2000)	Jang (2000) reported anions, cations, and metals leaching from individual CDD materials subjected to synthetic precipitation.
MSW-DST	This model's primary focus is the MSW stream and the environmental implications of its management. Wood is recognized as a major component of MSW; EOL LCI data for leachate, and LFG specific to branches are contained in the model.
Townsend et al. (1999, 2004, 2005)	These studies reported column and batch leaching tests data on new as well as weathered untreated and treated wood. Batch leaching tests utilized multiple set-up protocols to evaluate the impact on metal leachability.
US EPA (2012)	US EPA (2012) presents GHG emissions factors pertaining to source reduction, recycling, combustion, and disposal of dimensional lumber, MDF, and hardwood flooring for WARM model.
Athena	The Athena Impact Estimator (IE) for Buildings life-cycle model includes the energy requirement for demolishing wood-framed structures. The Athena Sustainable Materials Institute (ASMI) has developed cradle-to-gate LCI for the following wood products: cross laminated timber (CLT), Glulam (glue laminated timber), wood I-joists, laminated veneer lumber, MDF, OSB, particle board, softwood plywood sheathing, and softwood lumber.
EASETECH	EASETECH presents the LCI data associated with windrow composting of source separated organic waste which may include several wood products. LCI data specific to composting wood and SSOs in general is contained in the composting processes.
Ecoinvent	Doka (2009) presents elemental compositions of wood, which are used in conjunction with transfer coefficients to estimate contaminants emissions to different media.
WRATE	WRATE presents the emissions data of various constituents associated with landfilling unspecified wood, wood packaging, and non-packaging wood.

6.4 LCI Related to Wood Products Manufacturing

As discussed in Section 6.2, the reuse of recovered CDD wood in new construction or renovation or for manufacturing new wood products is limited in the US. Assuming a constant wood products demand, the reuse of discarded CDD wood products would offset production and the associated emissions of the same wood product from primary inputs. This section discusses the cradle-to-gate wood products manufacturing process and the associated LCI. Through CORRIM, several US wood products manufacturers, researchers, associations, and government agencies have collaboratively developed LCI for multiple wood products manufacturing unit processes, including forest management, harvesting, and manufacturing for various geographical regions (Northwest, Southeast, Inland Northwest, and Northeast-North Central) in the US (Oneil et al 2010, Puettmann et al. 2010). The LCI data discussed in this section are based primarily on the work of CORRIM. The LCI developed by CORRIM are included in the US LCI and EPA databases.

The major operations for manufacturing wood products include forestry operation, timber harvesting, and processing timber at a mill. Forestry operation includes activities such as site preparation for planting, planting seedlings or promoting natural regeneration or sprouting, fertilization, thinning, and reducing wild fire hazards (Oneil et al. 2010). Plants take up, convert, and store the atmospheric carbon until harvested. Timber harvesting entails cutting the trees (felling); removing limbs and tops and cutting the tree into merchantable and transportable log lengths, moving logs from the felling point to a loading point near a haul road; and loading and transporting logs from the forest to a process point (e.g., mill) (Oneil et al. 2010).

The LCI associated with the individual unit processes of forestry and harvesting operations are included in the forestry and logging process category of the US LCI (2012) database. The LCI for two major wood types (softwood and hardwood), multiple geographic regions of the US, and various management intensities (low, medium, and high) are included in these databases. The management intensity is a measure of the level of undertaking and spending on forest/land management (Arano and Munn 2006). The forestry LCI include plant uptake of the atmospheric carbon dioxide.

The harvested timber is debarked (i.e., exterior bark is removed) to produce "roundwood." Roundwood is used to produce a variety of wood products. As discussed earlier, approximately 60 MMT of solid wood products were manufactured in the US in 2011. Figure 6-2 presents a relative distribution of different wood products manufactured in the US (Howard and Westby 2013). It can be seen that lumber (hardwood and softwood), OSB, softwood plywood, and particleboard constituted 58%, 10%, 7%, and 5% of the total wood products manufactured in the US in 2011, respectively. These four products represented 80% of the total solid wood products manufactured in the US in 2011. Other industrial products include poles, piling, fence posts, and cooperage logs (Howard and Westby 2013). These wood products can be broadly classified into two categories: sawn lumber and composite or engineered wood products. Sawn lumber (including poles, piling, fence posts) is a single wood piece, whereas engineered wood products are made from lumber, veneers, strands of wood, or from other small wood elements that are bonded together with structural resins to form lumber-like structural products (US EPA 1990).

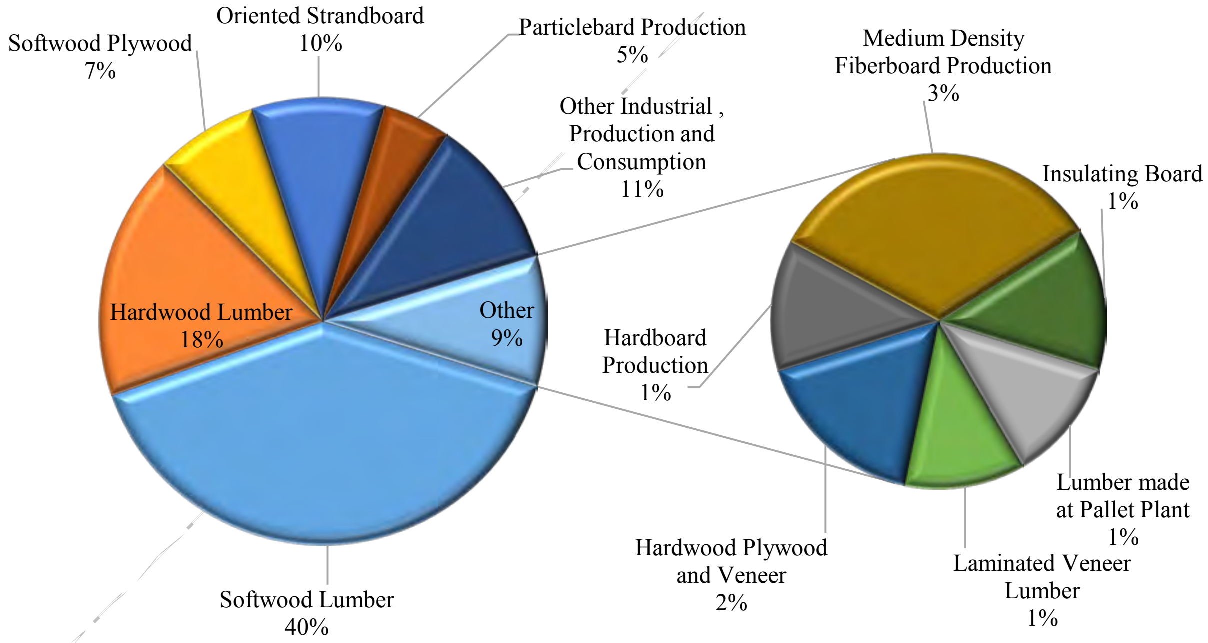

Figure 6-2. Distribution of Wood Products Manufactured in the US in 2011

Sawn lumber is manufactured by sawing the roundwood to produce "green lumber," which is dried in kilns to produce rough dry lumber. After drying, wood is shaped into final lumber form by planing rough dry lumber. As can be seen from Figure 6-2, plywood, OSB, particle board, and medium-density fiberboard are the most commonly manufactured engineered wood products. This section describes the manufacturing of these major engineered products based on process descriptions provided in the US EPA AP-42 documents.

For the manufacturing of softwood or hardwood plywood, debarked logs are cut to appropriate lengths and heated in hot water baths or via steam or hot spray or a combination of the three to around 93 °C. The heated logs are processed using a slicer or veneer lathe to generate veneer. The veneer is dried in kilns or driers. Formaldehyde-based resins are applied on both sides of the dried veneer using glue spreaders and covered with veneers with no glue. Multiple layers of veneers are laid together, with the grains of adjacent veneer layers perpendicular. The laid-up assembly is consolidated under heat and pressure in a hot press to press the glue into a thin layer over each veneer sheet and to activate thermosetting resins. The temperature and time depend on the wood species, the resin and the press design. The plywood is trimmed at the edges and the face and back may or may not be sanded smooth.

For OSB manufacturing, debarked logs are cut and placed in hot ponds (18-43 °C). The logs are the sliced into wafers (1.5 inches wide and 3 to 6 inches long) using a waferizer. The wafers may be passed through screens to remove fine and differentiate core and surface materials. Wafers are dried in rotary or conveyor driers, normally fired with wood residues from the plant. The dried wafers are processed to remove fines and segregate wafers by surface area and weight using a cyclone; undersized materials are used as fuel for the dryer burner or boiler. The dried wafers are blended with resin, wax, and other additives in a blender; thermosetting phenol-formaldehyde and isocyanate resins are the most commonly used binders. The resin-coated wafers are metered out on a moving screen. The wafers are mechanically oriented in one direction as they fall to the screen below. Wafers in the subsequent layer are oriented perpendicular to those in the previous layer. The continuous formed mat is cut into desired lengths. The trimmed mat is pressed under heat and pressure to activate the resign and bond the wafers. After cooling, bonded panels are trimmed to final dimensions, finished as necessary, and packaged.

MDF is typically made from wood chips (residues from other wood processing steps or from primary wood). The chips are cleaned and mechanically pulped to produce fibers. The fibers are blended with bonding resins (urea-formaldehyde is the most commonly used resin) and other additive. The resinated fibers are dried in single- or multi-stage dryers. The drying and blending sequence depends on the fibers-resins (along with other additives) blending method. The dried resinated fibers are deposited on a continuously moving belt to form a mat. The mats are prepressed and trimmed. The mat is pressed under heat and pressure to activate the resin and bond the fibers into solid boards. The boards are cooled, sanded, trimmed, and sawed to final dimensions; the boards may be painted or laminated as well.

Particleboard incorporates small particles typically in the form of a panel or other shapes. The source for the particles may be residues (e.g., wood shavings, sawdust) from other wood products manufacturing processes or harvested logs. After general size reduction (i.e., milling), the particles are screened and classified by size, using an air classifier. After the particle size equals the specifications, the particles are dried (at about 1600 °F for raw, greenwood particle inputs) to the desired moisture content (about 2 to 8% by mass). Screening may occur after this step for further fines removal. The particles are then combined with synthetic resin or other adhesive (e.g., wax) via spray nozzles. Wax may also be added to the boards' outer layers for protection. To form the solid end product, the resinated particles are formed into the desired shape. A press can be used to activate the resin and bond the fibers (for about 2.5 to 6 minutes) into boards. The boards are then cooled, sanded, and trimmed.

The cradle-to-gate materials and energy inputs and the emissions depend on the wood (softwood, hardwood) and product type (lumber, plywood, medium-density fiberboard) (Puettmann et al. 2010). Dimensional lumber processing involves only saw and planing mill operation (i.e., planed dried lumber), with the possible application of treatment chemicals (Wagner et al. 2009). Engineered wood products require a greater degree of processing and treatment (e.g., resin production/application and heating) than dimensional lumber products, often including the use of adhesives and chemicals. The manufacturing phase is reported to consume 90-92% of the total cradle-to-gate energy consumption for wood products manufacturing (Puettmann et al. 2010). The harvesting and material transport phases combined consume less than 10% of the total production energy. Drying has been reported to be the most energy-intensive step in lumber production (Puettmann and Wilson 2005). Drying and final pressing of composite products have been reported to be the most energy-intensive steps in engineered wood products manufacturing (Puettmann and Wilson 2005).

The US LCI (2012) and US EPA LCI databases provide multiple unit processes pertaining to softwood and hardwood lumber and various engineering products manufacturing; additional LCIs for wood product manufacturing were not developed as part of this project. In addition to US LCI and US EPA databases, the US EPA (1995a) reports air emissions [particulate matter (<2.5 and 10 microns), carbon monoxide, carbon dioxide, nitrous oxides and volatile organic compounds] from wood product manufacturing. GaBi contains US-specific LCI data for wood product reduction, which in large part overlap with US EPA and US LCI (2012) datasets; however, several unique process datasets were developed by PE International related to wood product manufacture (e.g., production of CCA containerboard (not a CDD wood product considered in this report)). The CCA production LCI dataset includes material inputs of chromic acid, lead, and copper sulfate to the process and electricity and steam use and transport for the US. Emissions to the environment are not provided within the accessible metadata for the dataset. The US EPA's WARM model presents GHG emission data pertaining to source reduction, recycling, and landfilling of dimensional lumber, MDF, and hardwood flooring from 100% primary inputs based on emissions from process and transport energy use and the elimination of forest carbon storage (due to the cessation of CO_2 sequestration by the trees when they are harvested). Athena (2012) provides Canada-specific LCI data for the manufacture of wood products. Several inputs used for developing these LCI are based on US LCI and CORRIM data.

6.5 LCI Related to Disposal

Disposal of CDD wood product waste in landfills is the most commonly encountered management practice in the US (Falk and McKeever 2012). Due to generally lower tipping fees, CDD wood disposal in MSW landfills is less common than disposal in CDD materials landfills. The potential for leachate and LFG release to the environment is dependent on the biogeochemical environment of the landfill and the environmental controls, as discussed in Chapter 2. The emissions associated with production and use of different materials and energy inputs for landfill construction, operation, and closure as well as those associated with leachate and gas should be included in LCI for wood disposal. The details of leachate and gas emission LCI are presented in this section. More details on landfill operation LCI are presented in Chapter 2.

Due to its organic nature, the decay of wood wastes in an anaerobic (i.e., oxygen-poor) environment produces methane, which may be collected by a GCCS and converted to biogenic CO_2 via flaring or energy-conversion technology. Most of the waste LCA models (WARM, MSW-DST, EASETECH) account for methane as the only gaseous emission associated with wood decomposition in a landfill. Moreover, WARM, and MSW-DST adopted material-specific methane yield and decay rates reported by the same sources (i.e., Eleazer et al. 1997, Barlaz 1998, Staley and Barlaz 2009, De la Cruz et al. 2010) for estimation of methane emissions. Eleazer et al. (1997) collected branches (<5 cm in length), from a compost facility in North Carolina and conducted bioassay tests in quadruplicate in 2-L reactors to estimate methane yield per unit of dry weight; no information on the tree species was provided.

The MSW-DST contains LCI data for leachate emissions from the landfill disposal of an array of waste materials contained in MSW, including branches, a component of yard waste. CDD wood product is not included as a material category in the model. The US EPA (2012) used methane yield of branches (reported on dry weight of branches) as a proxy for developing emission factors for dimensional lumber, medium-density fiberboard, and wood flooring. The methane yield was adjusted for moisture content to estimate methane emission per unit wet weight of the material. The moisture content used for dimensional lumber and medium density fiberboard was the same as that of branches. The moisture content used for wood flooring was greater than that of branches resulting in approximately 15% lower methane emission factors for wood floor than the other wood products. WRATE provides wood-specific emissions of approximately 30 gaseous constituents for disposal scenario. The materials-specific gaseous emissions of various constituents are not based on actual material-specific measurements, but rather are based on a theoretical allocation of the total emissions to individual waste components (Golder Associates 2005).

The LFG production properties of branches was used as a proxy for estimating gas generation as a result of the landfill disposal of CDD wood. Additional information on calculations used to estimate CDD wood LFG emissions is provided in Section 2.5.10.8. The methane and carbon dioxide emissions from the landfill disposal of CDD wood is respectively estimated as 0.064 and 0.21 kg for placement in a CDD landfill.

A significant fraction of methane is captured and combusted to carbon dioxide at the landfills with GCCS (e.g., MSW landfills). The methane and biogenic carbon dioxide emission from landfills with GCCS would be lower and higher, respectively, than from landfills with no GCCS. Using the average nationwide statistics for the percentage of landfills that have GCCS and assuming a 90% average gas collection efficiency as provided for dimensional lumber in US EPA (2012), the methane and carbon dioxide emissions from the landfill placement of one kilogram of wood in an MSW landfill are estimated to be 0.022 kg, and 0.33 kg, respectively. Wood treatment chemicals were assumed not to have an impact on gas generation.

Wood products are often treated with preservative chemicals for protection from the weathering elements and biota. Chemicals may either be applied to the wood's surface and/or impregnated (requires pressure treatment to infuse the chemical) into the wood itself (Haverty and Micales-Glaeser 2004, US EPA 1999). Chromated copper arsenate (CCA) was previously the most extensively used chemical to treat lumber and other wood products; CCA use began in the 1940s. Three types of CCA-treated wood were available. Type C CCA was the most common wood preservative when it was in widespread use (prior to phase-out) (Jambeck 2004). Other treatment chemicals, such as the copper-based alkaline copper quaternary (ACQ), copper azole (CBA), and disodium octaborate tetrahydrate (DOT) are being used in greater volume due to phase-out of CCA. Jambeck et al. (2007) estimated that the peak quantity of CCA wood in the waste stream would occur in 2008, when approximately 9.7 million m^3 would be disposed of. The presence of wood treatment chemicals and other chemicals in paints (which may contain lead), stains, etc. has a significant impact on the quality of liquid emissions (Lebow et al. 2004, Townsend et al. 2004) and complicates and often impedes the EOL management via disposal or reuse/recycling.

Lebow et al. (2004) conducted an extensive review of the published leaching data from treated wood and reported that several factors such as particle size, wood species, leaching water characteristics, and surface finishes (e.g., paint) have significant impact on preservative leaching from pressure-treated wood. For example, Lebow et al. (2004) reported that red oak leached approximately 15% of the total As, while yellow poplar leached only approximately 1% of As for the same treatment type (Type C CCA). Townsend et al. (2004) reported that for the same treated wood, a 100-g block leached at levels approximately a quarter of the arsenic leached by sawdust particles.

Similar to other CDD materials, wood is typically disposed of with other CDD materials. The actual measurement of long-term wood-specific liquids emissions from full-scale landfills are not available and not expected to be available in the future. The laboratory-scale leachate quality data published by various sources (Townsend et al. 1999, Jang 2000, Lebow et al. 2002, Lebow et al. 2004, Jambeck 2004, Dubey 2005, Jambeck et al. 2006, Dubey et al. 2007, Mitsuhashi et al. 2007, Dubey et al. 2010, Hasan et al. 2010, Clausen et al. 2010, Tao et al. 2013, Tao 2014) were reviewed to estimate liquids emission from the disposal of untreated and treated wood in landfills. The following criteria were used to select data for estimating liquids emission from wood products disposal in landfills:

1. **Sample size**. Wood products are not expected to undergo a significant size reduction during waste placement and compaction in a landfill. The data from leaching tests conducted on larger particle sizes (e.g., wood blocks) were preferred over data from tests on aggressively size-reduced wood samples (e.g., sawdust) (e.g. Townsend et al. 2004, and Townsend et al. 2005) for liquids emission estimates.

2. **L:S ratio**. The cumulative amount of chemicals leached from treated wood has been reported to be a function of the amount of liquid wood is exposed to (Jambeck 2004, Tao et al. 2013). Many studies assessed leaching from sample columns exposed to natural or synthetic precipitation (Jang 2000, Jambeck 2004, Tao et al. 2013, and Tao 2014) for a limited timeframe. None of these studies reported 100% leaching of the preservatives. The L:S ratio for these studies was either not reported or significantly lower than L:S ratio of the standardized leaching tests such as SPLP and TCLP. The data from tests with greater L:S ratio were preferred over those from lower L:S ratio tests.

3. **Leaching fluid**. As leaching in landfill environment is expected to occur under slightly acidic conditions, leaching data associated with neutral or basic fluids such as deionized water (e.g., Jambeck 2004, Jambeck et al. 2006, Dubey et al. 2007) were not used for estimating liquids emission associated with disposal of untreated/treated wood in landfills. Data from tests using SPLP and TCLP extraction fluids were used for estimating liquids emission from an unlined inert debris landfill and MSW landfill, respectively.

Based on these criteria, results from SPLP batch leaching test (L:S=20) data reported by Jang (2000) were used for estimating liquids emission (for COD, chloride, potassium, calcium, arsenic, chromium, copper, and manganese) from untreated wood and CCA wood disposal in an unlined inert debris (CDD materials) landfill. Jang (2000) conducted batch leaching tests using SPLP extraction fluid on individual CDD materials, including untreated and CCA-treated wood. The column leaching data from Townsend et al. (1999) were used for parameters that were measured below the detection limit by Jang (2000). Townsend et al. (1999) conducted column leaching tests on individual CDD materials, including wood (untreated, new, southern pine lumber) with SPLP extraction fluid. The overall L:S ratio for the column experiment was approximately 5.

Table 6-3Table 6-3 presents a proposed LCI for untreated wood disposal in unlined CDD materials landfill. The bulk density of wood waste presented in CCG (2006) was used to estimate cover soil requirements for placement of untreated wood waste at a CDD landfill. Additional details on the diesel and electricity requirements included in the "CDD landfill operations" flow and details on the cover soil estimate are provided in Chapter 2. A 20 km average nationwide distance between the wood removal site and CDD landfills was assumed.

Table 6-3. Proposed LCI Dataset: Untreated Wood Waste, at Unlined CDD Landfill

Input Flow	**Source**	**Category**	**Units**	**Amount**
Untreated wood waste, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.176
Output Flow	**Source**	**Category**	**Units**	**Amount**
Untreated wood waste, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.064
Carbon dioxide	US EPA (2012)	Air/Unspecified	kg	0.21
Chloride	Jang (2000)	Water/groundwater	mg	74.0
Calcium	Jang (2000)	Water/groundwater	mg	24.0
COD	Jang (2000)	Water/groundwater	mg	2,400
Potassium	Jang (2000)	Water/groundwater	mg	42.0
Manganese	Jang (2000)	Water/groundwater	mg	2.20
Magnesium	Townsend et al. (1999)	Water/groundwater	mg	29.0
Carbonate	Townsend et al. (1999)	Water/groundwater	mg	6.1
Sodium	Townsend et al. (1999)	Water/groundwater	mg	7.5

Leaching of chemicals from wood treated with CCA and other chemicals have been investigated by several authors (e.g., Jang 2000, Townsend et al. 2004, Jambeck 2004, Dubey et al. 2010, Hasan et al. 2010). Based on the criteria discussed above, data presented by Jang (2000) were selected to estimate liquid emissions associated with CCA-wood (Type C, chemical retention rate of 4.0 kg/m^3) disposal in unlined CDD materials landfill. Data from SPLP tests conducted by Dubey et al. (2010) on sawdust of CCA-treated wood (Type C, retention rate of 6.4 kg/m^3) were used for the parameters not measured by Jang (2000). Table 6-4 presents proposed LCI for CCA wood disposal in unlined CDD materials landfill. As can be seen from Table 6-4 COD, chloride, calcium, manganese, and potassium emission from CCA-treated wood are at similar levels to those from untreated wood.

Table 6-4. Proposed LCI Dataset: CCA-Treated Wood Products, at Unlined CDD Landfill

Input Flow	Source	Category	Units	Amount
CCA-treated wood, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.176
Output Flow	**Source**	**Category**	**Units**	**Amount**
CCA-treated wood products, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.064
Carbon dioxide	US EPA (2012)	Air/Unspecified	kg	0.21
Arsenic	Jang (2000)	Water/groundwater	mg	47.6
Boron	Dubey et al. (2010)	Water/groundwater	mg	9.0
Chromium	Jang (2000)	Water/groundwater	mg	19.4
Copper	Jang (2000)	Water/groundwater	mg	10.4
COD	Jang (2000)	Water/groundwater	mg	2,600
Chloride	Jang (2000)	Water/groundwater	mg	72.0
Calcium	Jang (2000)	Water/groundwater	mg	50.0
Potassium	Jang (2000)	Water/groundwater	mg	82.0
Manganese	Jang (2000)	Water/groundwater	mg	2.8

Table 6-5, 6-6, and 6-7 present proposed LCI for disposal of ACQ-, CBA-, and DOT-treated wood in unlined CDD landfills. The liquids emissions for these treated wood types were estimated based on SPLP data presented by Dubey et al. (2010). Dubey et al. (2010) conducted tests on sawdust from various treated wood using a variety of extraction fluids, including SPLP, TCLP, and leachates from MSW landfills. The same quantity of cover soil, diesel and electricity consumption for landfill operations, and transport distance between the site of wood removal and the CDD landfill were assumed as was assumed for the CDD landfill disposal of untreated wood.

Table 6-5. Proposed LCI Dataset: ACQ-Treated Wood Products, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
ACQ-treated wood, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.176
Output Flow	**Source**	**Category**	**Unit**	**Amount**
ACQ-treated wood products, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.064

Carbon dioxide	US EPA (2012)	Air/Unspecified	kg	0.21
Arsenic	Dubey et al. (2010)	Water/Groundwater	mg	0.55
Boron	Dubey et al. (2010)	Water/Groundwater	mg	168
Copper	Dubey et al. (2010	Water/Groundwater	mg	391
Chromium	Dubey et al. (2010)	Water/Groundwater	mg	1.05

Table 6-6. Proposed LCI Dataset: CBA-Treated Wood Products, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
CBA-treated wood, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.176
Output Flow	**Source**	**Category**	**Unit**	**Amount**
CBA-treated wood products, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.064
Carbon dioxide	US EPA (2012)	Air/Unspecified	kg	0.21
Arsenic	Dubey et al. (2010)	Water/Groundwater	mg	0.1
Boron	Dubey et al. (2010)	Water/Groundwater	mg	341
Copper	Dubey et al. (2010	Water/Groundwater	mg	619
Chromium	Dubey et al. (2010)	Water/Groundwater	mg	0.3

Table 6-7. Proposed LCI Dataset: DOT-Treated Wood Products, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
DOT-treated wood, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.176
Output Flow	**Source**	**Category**	**Unit**	**Amount**
DOT-treated wood products, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.064
Carbon dioxide	US EPA (2012)	Air/Unspecified	kg	0.21
Arsenic	Dubey et al. (2010)	Water/Groundwater	mg	2.28
Boron	Dubey et al. (2010)	Water/Groundwater	mg	1,450
Copper	Dubey et al. (2010	Water/Groundwater	mg	1.32
Chromium	Dubey et al. (2010)	Water/Groundwater	mg	0.115

Leachate related emissions from wood products in an MSW landfill environment were estimated based on TCLP data reported in literature. Table 6-8 through Table 6-11 present the proposed LCI for four types of treated wood based on the TCLP test data reported by Dubey et al. (2010). The cover soil requirements and material and energy flows included for MSW landfill construction, operation and closure and post-closure care are detailed in Chapter 2 of the report. The bulk density of wood products presented in CCG (2006) was used to estimate cover soil requirements. The distance between the wood removal site and MSW landfills was assumed to be 20 km in the absence of average nationwide data.

Table 6-8. Proposed LCI Dataset: CCA-Treated Wood Products, at MSW Landfill

Input Flow	Source	Category	Units	Amount
CCA-treated wood, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	1.24
Output Flow	**Source**	**Category**	**Units**	**Amount**
CCA-treated wood products, at an MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.022
Carbon Dioxide	US EPA (2012)	Air/Unspecified	kg	0.33
Arsenic	Dubey et al. (2010)	Water/groundwater	mg	226
Boron	Dubey et al. (2010)	Water/groundwater	mg	12.6
Chromium	Dubey et al. (2010)	Water/groundwater	mg	74.7
Copper	Dubey et al. (2010)	Water/groundwater	mg	217

Table 6-9. Proposed LCI Dataset: ACQ-Treated Wood Products, at MSW Landfill

Input Flow	Source	Category	Units	Amount
ACQ-treated wood products, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	1.24

Output Flow	Source	Category	Units	Amount
ACQ-treated wood products, at an MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.022
Carbon Dioxide	US EPA (2012)	Air/Unspecified	kg	0.33
Arsenic	Dubey et al. (2010)	Water/groundwater	mg	1.34
Boron	Dubey et al. (2010)	Water/groundwater	mg	181
Copper	Dubey et al. (2010)	Water/groundwater	mg	940
Chromium	Dubey et al. (2010)	Water/groundwater	mg	1.0

Table 6-10. Proposed LCI Dataset: CBA-Treated Wood Products, at MSW Landfill

Input Flow	Source	Category	Units	Amount
CBA-treated wood products, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	1.24
Output Flow	**Source**	**Category**	**Units**	**Amount**
CBA-treated wood products, at an MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.022
Carbon Dioxide	US EPA (2012)	Air/Unspecified	kg	0.33
Arsenic	Dubey et al. (2010)	Water/groundwater	mg	0.133
Boron	Dubey et al. (2010)	Water/groundwater	mg	393
Copper	Dubey et al. (2010)	Water/groundwater	mg	721
Chromium	Dubey et al. (2010)	Water/groundwater	mg	0.313

Table 6-11. Proposed LCI Dataset: DOT-Treated Wood Products, at MSW Landfill

Input Flow	Source	Category	Units	Amount
DOT-treated wood products, from EOL removal		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	1.24
Output Flow	**Source**	**Category**	**Units**	**Amount**
DOT-treated wood products, at an MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.022
Carbon Dioxide	US EPA (2012)	Air/Unspecified	kg	0.33
Arsenic	Dubey et al. (2010)	Water/groundwater	mg	2.84
Boron	Dubey et al. (2010)	Water/groundwater	mg	1,300
Copper	Dubey et al. (2010)	Water/groundwater	mg	3.35
Chromium	Dubey et al. (2010)	Water/groundwater	mg	0.280

Several factors should be considered when using the proposed liquids emissions for MSW landfills. First, the emissions presented in Table 6-8 through Table 6-11 should be considered as partial as these are based on batch leaching tests with an L:S ratio of 20. In reality, wood placed in a landfill would be subjected to leaching a much greater L:S ratio (assuming that the landfilled waste will never be reclaimed) associated with untreated leachate. Second, leachate from an MSW landfill is typically collected and treated before the effluent is discharged into the environment during active disposal, closure, and post-closure care. The wastewater treatment process partitions contaminants from the liquid phase into the treated effluent and solid residues (sludge, biosolids). The contaminant amounts released into the environment with effluent discharge depends on the treatment plant's contaminant-removal efficiency, which in turn depends on the contaminant type (NCSU and ERG 2011). A treatment efficiency of 85% for heavy metals is used by MSW-DST. The sludge from a wastewater treatment plant is commonly managed by either land-application or disposal at MSW landfills. Based on a nationwide survey NEBRA (2007) estimated that approximately 49% of the WWTP sludge generated in the US is land applied. The inorganic contaminants contained in sludge would potentially leach and be released into the environment (groundwater, surface water).

The additional pathways, apart from effluent discharge, by which leachate emits contaminants into the environment are fugitive leachate emissions through the bottom liner imperfections (e.g., geomembrane pin holes that occur during construction), leaching of chemicals from land-applied sludge, and the cyclic process of contaminant release from wastewater treatment plant sludge deposited in landfills, leachate treatment, and sludge disposal at landfills. Moreover, 100% of the leachate would discharge into the environment at the conclusion of the post-closure care period. For example, MSW-DST and EASETECH account for leachate collection and treatment for a default period of 100 years. EASETECH and MSW-DST documentations do not appear to consider contaminants emission from sludge disposal or sludge land application.

6.6 LCI Related to Recycling

Mulch and biomass fuel production are the two primary recycling options for CDD wood (Wiltsee 1998, Townsend et al. 2003). As discussed earlier, approximately 30% of wood (equivalent to more than 8 MMT) is recycled as mulch or biomass fuel (Wiltsee 1998, US EPA 2014). A lack of demand for end-products (mulch, biomass), coupled with competition from wood product manufacturing industries, are potential challenges for CDD wood recovery and recycling. The US mulch demand in 2005 of approximately 3 MMT (estimated by Cochran 2006) was small compared to the amount of residue produced by wood product manufacturing (177 MMT in 2002 as reported by McKeever (2004). Moreover, CDD wood is often considered to be a less-desirable feedstock for mulch production due to aesthetics (Townsend et al. 2003). Information on CDD wood composting is lacking, likely owing to the limited nature of this practice. All potential recycling options (including closed-loop recycling for wood product manufacturing and composting) entail wood processing (i.e., grinding/chipping) as the first step. As closed-loop recycling of discarded wood is not prevalent, processing and reuse of discarded wood for wood product manufacturing is not discussed further.

Energy inputs and emissions associated with CDD wood waste processing to produce mulch include those related to manufacturing and the use of sorting, grinding, and screening equipment (Cochran 2006). Based on the data reported by Morbark (2006), Diamond Z (2006), and Bandi (2006) (as reported by Cochran 2006) for a horizontal grinder and manufacturer equipment specifications (for an excavator and loader), the diesel equipment energy requirements is 29.5 MJ per MT of wood waste processed in a mixed CDD MRF. This is equivalent to a fuel consumption of approximately 0.755 L of diesel per MT of wood. As a point of comparison, Levis (2008) estimated yard-waste shredding fuel consumption to be approximately 1.18 L and 3.0 L per MT; the estimate was based on a regression analysis of production rate, horsepower, and fuel consumption data from manufacturers for several models of horizontal grinders and tub grinders, respectively. This wide range of data suggests a need for measuring energy and material inputs from actual facility operations so that more reliable LCI can be developed. Until such data are available, the use of the fuel consumption data reported by Cochran (2006) is proposed.

The non-energy-related emissions from wood grinding include particulate matter emission and liquid emission from wood/wood chip stockpiles. AP-42 presents air emission factors for log chipping as part of MDF manufacturing. These data can be used as a proxy for CDD wood grinding until measurements from operating facilities become available. However, it appears that unlike CDD wood processing facilities, engineering controls such as cyclone and/or fabric filter collection are implemented to control particulate matter emission from chipping operations at MDF manufacturing facilities. Using log chipping air emission as a proxy would, therefore, result in underestimating particulate matter emission from CDD wood processing facilities. As the wood decomposition in this scenario would occur under aerobic conditions, gas emissions from the land application of mulch were estimated by assuming that 100% of carbon content will decompose to produce carbon dioxide. Using the biogenic carbon content of branches (published by Barlaz 1998) as a proxy for wood products, approximately 1.63 kg of carbon dioxide (biogenic) would be produced from aerobic decomposition of 1 kg of wood products; this estimate is based on 0.494 g of carbon content as C per dry kg of wood product and 0.9 kg of dry wood per kg of wet wood product.

The liquids emissions from land application of mulch are expected to be the same as those from wood disposal in CDD materials landfills as leaching is primarily influenced by natural precipitation. It was assumed that the CDD wood would be transported 20 km from the job site to the processing facility and the mulch would be transported 20 km to the mulch end user. Table 6-12 presents proposed LCI for CDD wood grinding to produce mulch and land application of mulch.

Table 6-12. Proposed LCI Dataset: Mulch Production and Land Application

Input Flow	Source	Category	Units	Amount
Wood waste		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*40
Diesel, combusted in industrial equipment	Cochran (2006)	Flows	L	0.000755
Output Flow	**Source**	**Category**	**Units**	**Amount**
Land-applied mulch		*Construction and Demolition Debris Management*	*kg*	*1*
Carbon Dioxide (biogenic)	US EPA (2012)	Air/Unspecified	kg	1.63
Chloride	Jang (2000)	Water/groundwater	mg	74.0
Calcium	Jang (2000)	Water/groundwater	mg	24.0
COD	Jang (2000)	Water/groundwater	mg	2,400
Potassium	Jang (2000)	Water/groundwater	mg	42.0
Manganese	Jang (2000)	Water/groundwater	mg	2.20
Magnesium	Townsend et al. (1999)	Water/groundwater	mg	29.0
Carbonate	Townsend et al. (1999)	Water/groundwater	mg	6.1
Sodium	Townsend et al. (1999)	Water/groundwater	mg	7.5

6.7 LCI Related to Combustion

As described earlier, one of the predominant markets for mixed wood recovered at CDD processing facilities is use as a boiler fuel. Cochran et al. (2006) estimates that approximately 5 MMT of waste wood products could be combusted for electricity production without expanding the US capacity for wood combustion. Recovered wood is typically size reduced at these facilities and transported for combustion in incinerators or boilers (McKeever 2002). At these combustion facilities, the CDD debris is often mixed with other sources of woody debris, such as yard trash, LCD, pulp and paper mill residues, and agriculture/silviculture waste. Wood wastes may also be dried to concurrently reduce their moisture content and raise their heating value. However, for use as a boiler fuel, this drying step appears to not be necessary for the relatively low moisture contents (e.g., 10 – 20%) commonly encountered in CDD wood products (Koch 1980, US EPA 1995b). Drying is more of a concern for treating green wood (i.e., freshly cut) residue feedstock from the timber industry; Curkeet (2011) estimates that green wood may have a moisture content greater than 75%.

The environmental emissions produced from combustion will depend on the boiler configuration in combination with in-place environmental controls. Different types of boiler configurations may be used, such as spreader stoker, Dutch oven, and suspension-fired, each of which may release a different set of emissions (e.g., fluidized bed combustion reduces the emissions associated with incomplete combustion byproducts) (US EPA 1995b). US-specific LCI data related to wood waste combustion as part of an MSW mass burn scenario is included in the US EPA WARM model, where the focus is on MSW mass burn because of the relatively low number of MSW RDF fuel facilities in the US. The biogenic carbon dioxide released as a result of wood combustion is not accounted for in WARM; however, a nitrous oxide emission factor is assigned to the wood waste material categories (i.e., dimensional lumber, wood flooring, MDF).

The nitrous oxide emission factor is not wood specific – it is evenly allocated to all possible nitrogen-containing waste materials based on the results of mixed MSW combustor emissions reported by the Intergovernmental Panel on Climate Change (IPCC) (US EPA 2012).

The US LCI (2012) database contains emissions data for two wood-combustion processes (i.e., combustion of wet and dry wood residue) based on US EPA (1995b) AP-42 emissions included in the database's "Other Electric Power Generation" subfolder of the "Utilities" folder. An additional process located in the same place represent electricity generation at a biomass plant. However, the process system boundaries include biomass production, and thus emissions from biomass growth/management and combustion cannot be distinguished. Additional wood combustion LCI datasets can be found in the "Steam and Air-Conditioning Supply" subfolder of the "Utilities" folder, which include another 14 process datasets that simulate the combustion of different wood fuels combusted in industrial boilers. These processes are subcategorized by whether they handle softwood or hardwood, the region of the US the wood originated from, and the specific type of facility which produced the wood material.

Since emissions for the combustion of dry wood residue have already been compiled from AP-42 into an LCI process dataset, the use of this dataset to model air emissions released from the combustion of processed CDD wood is proposed. However, to build a product system that incorporates this existing process, the process would need to be modified to have a processed wood input flow and would need to have an output flow of combusted wood ash. Currently, the emissions from the "Combustion, wet wood residue, AP-42" process are quantified per energy (MJ) recovered from the combustion process. AP-42 (US EPA 1995b) and US EPA (2012) together support a moisture content of CDD wood (e.g. dimensional lumber, furniture) of 10% (wet basis) and Jenkins et al. (1998) presents the ash content of demolition wood as 13% of initial dry mass; together, these statistics could be used to update this process so that it could be included with a CDD wood combustion product system.

6.7.1 Wood Ash

Wood ash is generated at a rate of approximately 2.7 MMT annually in the US (Risse 2010). In the US, ash is managed by two major pathways: landfilling (approx. 65%) and land application (9%), while 25% goes towards "other" undefined uses (Spokas 2010). When used as a soil amendment, wood ash is capable of providing valuable nutrients (e.g., potassium, phosphorus, magnesium) as well as acting as a liming agent, raising the pH and thus assisting in the retention of nutrients (Kahl et al. 1996, NEWMOA 2001, ASTSWMO 2007, ODEQ 2011). Land application is practiced more frequently in the northeastern US (at a rate of approximately 80% generation); in contrast, the Southeastern US practices land application at only about 10% and the Midwest at about 33% (Vance 1996, Risse 2010). Although practiced on a more limited scale, wood ash may also be composted with sewage sludge, practiced at a rate of about 5% in the northeastern US (Spokas 2010). Additional uses for this material include as an ingredient in concrete manufacture (due to potential pozzolanic properties) and as alternative daily or intermediate cover in landfills (Naik 2001, ODEQ 2011).

The presence of CCA-treated wood has been recognized as a major issue with CDD wood waste combustion (Cochran 2006). Incinerated CCA-treated wood can produce ash with heavy metal concentrations that exceed toxicity characteristic hazardous waste limits (Solo-Gabriele et al. 2002). In batch leaching tests (TCLP, SPLP) performed by Solo-Gabriele et al. (2002), wood ash produced from mixed wood waste with only 5% CCA-treated wood (by mass) caused consistent exceedance of toxicity limits for arsenic and intermittent exceedances for chromium. For ash resulting from the combustion of CCA-treated wood retaining high levels of preservative, heavy metals represented up to 36% (by weight) of the resulting ash. Regulations related to beneficial use of wood ash reflect concerns over the presence of treated wood combustion ash. For example, Florida allows the land application of wood ash provided it was not produced from combustion of treated or painted wood (FDEP 2002).

Wood ash composition and characteristics (e.g., leaching behavior, reactivity) can vary significantly, depending on the temperature of the system and the characteristics and degree of contaminants in the fuel wood (Jenkins et al. 1998). Several studies have examined the chemical characteristics and the wood ash effects on plant growth as well as total metals extractable under variably acidic conditions (Zhan et al. 1994, Demeyer et al. 2001, Norstrom et al. 2012). Other researchers in the US have examined ash leaching using deionized water, ammonium citrate, and humic and fulvic acids as extraction fluids (Erich 1991, Clapham and Zibilske 1992, Chirenje et al. 2002). Except for the limited testing performed by Solo-Gabriele et al. (2002), leaching data from wood ash in conditions simulating precipitation exposure or a landfill environment were not found in the literature.

Table 6-13 through Table 6-17 present the proposed LCI datasets for untreated or treated wood (provided at different treatment levels) ash in both CDD and MSW landfill environments based on the data reported by Solo-Gabriele et al. (2002); the SPLP results (used to simulate wood ash placed in a CDD landfill) and TCLP results (used to simulate wood ash placed in an MSW landfill) were derived from leaching tests performed on seven ash samples. It was assumed that treated-wood-derived ash would be disposed of in a lined cell (MSW landfill) and not in an unlined landfill (CDD landfill). These samples included ash from untreated wood (southern yellow pine) (one sample), ash from pre-consumer CCA-treated wood (three samples, each with a different treatment level), and ash produced from recycled wood recovered from CDD processing facilities (three samples, each recovered from a different facility). The SPLP and TCLP leaching results from the ash samples produced from the combustion of the recycled wood recovered from the three CDD processing facilities were each averaged to respectively provide the leaching LCI data presented in Table 6-15 through Table 6-17. Only copper, chromium, and arsenic leaching results were provided in this study. The expected leachable concentrations of non-metal organics and inorganics are unknown; however, the total and leachable ash concentrations of these parameters as published in other studies (as listed above) suggest that detectable concentrations of numerous other parameters (e.g. calcium, manganese, iron) may be encountered if analyzed. Wood ash derived from untreated CDD wood did not show detectable leached concentrations of any of the three metals. The only environmental burdens included in the proposed LCI which represents the placement of untreated wood at a CDD or MSW landfill would be those associated with landfill construction, operations, and closure/post-closure care. The liquids emissions presented for MSW landfill disposal represent the emissions with untreated leachate. Leachate from lined landfills is typically collected and treated during active landfill operation, closure, and post-closure care prior to discharge into the environment.

The proposed datasets for placing ash in a CDD landfill may be used to simulate the land application of wood ash if the flows associated with landfilling and cover soil are removed. Except for untreated wood ash, all contaminants measurements below the detection limit were analyzed at the detection limit for dataset development purposes. The nationwide average distance between wood combustion and CDD/MSW landfills was not found; a distance of 20 km was assumed. For estimating the amount of landfill cover soil required, a bulk wood ash density of 702 kg/m^3 was assumed (Huang et al. 1992).

Table 6-13. Proposed LCI Dataset: Untreated Waste Wood Ash, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
Wood ash, from combustion		Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.0271
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Untreated waste wood ash, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*

Table 6-14. Proposed LCI Dataset: Untreated Waste Wood Ash, at MSW Landfill

Input Flow	Source	Category	Unit	Amount
Wood ash, from combustion		Construction and Demolition Debris Management	kg	1
Cover soil, at MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	0.190
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Untreated waste wood ash, at MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*

Table 6-15. Proposed LCI Dataset: CCA-Treated Wood Ash, 4 kg/m^3 CCA Retention Level, at MSW Landfill

Input Flow	Source	Category	Unit	Amount
Wood ash, from combustion		Construction and Demolition Debris Management	kg	1
Cover soil, at MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	0.190
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1

MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Arsenic	Solo-Gabriele et al. (2002)	water/groundwater	mg	1010
Chromium	Solo-Gabriele et al. (2002)	water/groundwater	mg	120
Copper	Solo-Gabriele et al. (2002)	water/groundwater	mg	11
Wood ash, CCA 4 kg/m^3, at MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*

Table 6-16. Proposed LCI Dataset: CCA-Treated Wood Ash, 9.6 kg/m^3 CCA Retention Level, at MSW Landfill

Input Flow	Source	Category	Unit	Amount
Wood ash, from combustion		Construction and Demolition Debris Management	kg	1
Cover soil, at MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	0.190
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Arsenic	Solo-Gabriele et al. (2002)	water/groundwater	mg	2660
Chromium	Solo-Gabriele et al. (2002)	water/groundwater	mg	2
Copper	Solo-Gabriele et al. (2002)	water/groundwater	mg	296
Wood ash, CCA 9.6 kg/m^3, at MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*

Table 6-17. Proposed LCI Dataset: CCA-Treated Wood Ash, 40 kg/m^3 CCA Retention Level, at MSW Landfill

Input Flow	Source	Category	Unit	Amount
Wood ash, from combustion		Construction and Demolition Debris Management	kg	1
Cover soil, at MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	0.190
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Arsenic	Solo-Gabriele et al. (2002)	water/groundwater	mg	10200
Chromium	Solo-Gabriele et al. (2002)	water/groundwater	mg	37.6
Copper	Solo-Gabriele et al. (2002)	water/groundwater	mg	236
Wood ash, CCA 40 kg/m^3, at MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*

6.8 Data Gap analysis and Opportunities for Additional LCI Data

Table 6-18 summarizes the type of data presented by various sources reviewed for compilation of wood products EOL management LCI. Several LCA models (e.g., WARM, MSW-DST) and LCI databases (US LCI, US EPA LCI) provide data with respect to US-based processes. Some sources used data from the other sources. For example, the US EPA (2012) used data from the NREL database and Cochran (2006). As shown in the table, many sources present only part of the data/information needed for LCI compilation. For example, WARM presents only GHG emissions and uses emissions only associated fuel consumption in equipment to estimate landfill emission factor. Similarly, Ecoinvent only has partial landfill leachate emissions data because leachate from inert materials landfills are not considered.

A majority of LCI information available on wood products pertains to the manufacturing aspects of the life cycle. Only limited EOL-specific LCI are available. Based on a review of the available information, the following data gaps were identified that, if collected, would allow for a more comprehensive LCI dataset for wood products EOL management:

1. **Data pertaining to CDD wood EOL management practices.** Only a few studies that attempted to assess CDD wood EOL management practices were identified; these studies estimated wood management practices based on either verbal survey or material flow analysis. As these data are of interest to multiple governmental agencies (e.g., US EPA, USDA, state environmental agencies), an opportunity for collaborative research exists to quantify current practices of wood management in the EOL phase.

2. **Long-term leachable emissions from wood products placed in a landfill.** As described earlier, the liquid emissions presented in this study are based on SPLP and TCLP tests, which simulate leaching from synthetic rainwater or an aggressive MSW landfill environment, respectively. Moreover, the batch leaching data used for estimating liquid emissions correspond to an L:S ratio of 20 and, therefore, do not represent complete liquid emission. In addition, leaching data of only certain chemical constituents are reported in literature. For example, studies evaluating leaching from CCA wood primarily reported copper, chromium, and arsenic data. Furthermore, the standardized leaching tests simulate leaching associated with physical and chemical mechanisms and do not simulate leaching associated with biological decomposition of wood due to the short duration of these tests (18 hours). As wood products are typically disposed of with other discarded materials and not disposed of in a monofill, field-scale leachate quality data specific to wood products disposal in landfill are rare and probably would not be available in the future. The liquids emissions from wood products placement in landfills would therefore need to be based on laboratory-scale studies. Future research should consider assessment of leaching of a larger suite of chemicals over a greater L:S ratio and those associated with biological decomposition.

3. **Long-term gaseous emissions from wood biodegradation in landfill**. The data reported for branches were used as a proxy for estimating gaseous emission from anaerobic biodegradation of wood production disposed of in landfills due to lack of CDD wood-specific data. Moreover, the emission of two compounds (methane and carbon dioxide) are included in the proposed LCI for landfill disposal of wood products. Future research should consider quantification of a larger suite of gaseous emission from dimensional as well as different engineered wood products to assess the impact of resins on gaseous emissions.

4. **Materials and energy input and emission from discarded wood products processing.** The wood products processing LCI presented in this report are based on the energy requirements reported by one data source (Cochran 2006). Cochran (2006) identified the energy requirement of CDD wood processing based on a survey of a few equipment manufacturers. Some of the data (e.g., consumables, fuel and electricity usage, water consumption, material throughput) tracked by the facility owner from a financial accounting perspective can be readily used for developing more comprehensive LCI for wood processing. The process non-energy emission (e.g., particulate matter emission into the atmosphere and liquids emission from short-term wood stockpile to water) are not included in the proposed LCI due to the lack of these data. Future research should consider collecting and compiling these data.

5. **Wood combustion ash.** Numerous studies characterized wood residue ash to assess its benefits as a soil amendment (e.g., as lime substitute). Leaching data (SPLP, TCLP), however, are lacking to assess liquids emission for untreated wood ash land-application or disposal scenarios. Only limited leaching data are available for CCA-treated wood ash. Future research should consider quantifying leaching emission of a wider suite of chemicals and for greater L:S ratio from wood ash (untreated as well as treated) for land application and various disposal scenarios.

6. **Environmental Impact of emerging wood preservation chemicals**. Although the phase out of certain treatment chemicals (CCA) has occurred, future research efforts should consider assessing the impacts of emerging treatment preservatives such as nano-zinc oxide (Clausen et al. 2010).

Table 6-18. Overview of Wood Product Specific LCI Data Available

Process	AP-42	Cochran (2006)	US EPA WARM	MSW-DST	US EPA/ NREL	Jang (2000)	Dubey et al. (2010)	Solo-Gabriele et al. (2001)
Dimensional Lumber manufacturing					X			
WPM- Engineered Wood Production	X				X			
WPM- Ancillary Materials (e.g., CCA)					P			
EOL Product Removal								
Landfill Construction & Operation			P	X				
Landfill Leachate Emissions						X	X	
Landfill Gas Emissions		X		X				
Wood Processing		X						
Mulch-Liquids Emission from Land Application						X		
Untreated Wood Combustion-Air Emission	X		P					
Treated Wood Combustion-Air Emission								
Untreated Wood Combustion Ash-Liquids Emissions								
Treated Wood Combustion Ash- Liquids Emission								X

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7 Land Clearing Debris

7.1 Introduction

LCD is comprised of tree tops, branches, and stumps and can also include materials such as soil, rocks, and shrubs resulting from vegetation removal for building/infrastructure construction and land development (US EPA 2011). According to a study conducted in Canada, approximately 97% of non-merchantable LCD is comprised of non-foliage woody material (PGEC 1997). The moisture content of LCD is typically greater than that of other wood sources (e.g., dimensional lumber, plywood) in CDD, estimated at approximately 50% (wet basis) (Maker 1994, US EPA 2003, Tumuluru et al. 2011).

Estimates of LCD production are difficult because this category of materials is often excluded from regulation as a solid waste. US EPA (2014) found that states representing approximately 65% of the U.S. population either did not include LCD in the definition of CDD or exempted LCD from solid waste regulations altogether. Wiltsee (1998) indicated difficulty with estimating LCD quantities because the major management approaches for LCD (which includes chipping on-site and burning without energy recovery) often do not involve any mass or volume estimates.

Figure 7-1 identifies the flow of materials and processes that should be considered for conducting an LCA of LCD EOL management. LCD is most commonly disposed of onsite (at the site of generation) through burning, but also may be disposed of offsite at a landfill (Wiltsee 1998). LCD is typically processed (e.g. chipped, ground-up, screened) prior to use in a recycling applications such as mulch production, compost, or combustion with energy recovery. Although wood in LCD can be used for wood product manufacturing, this is not a common practice. This EOL management option of LCD is, therefore, not discussed further in this report.

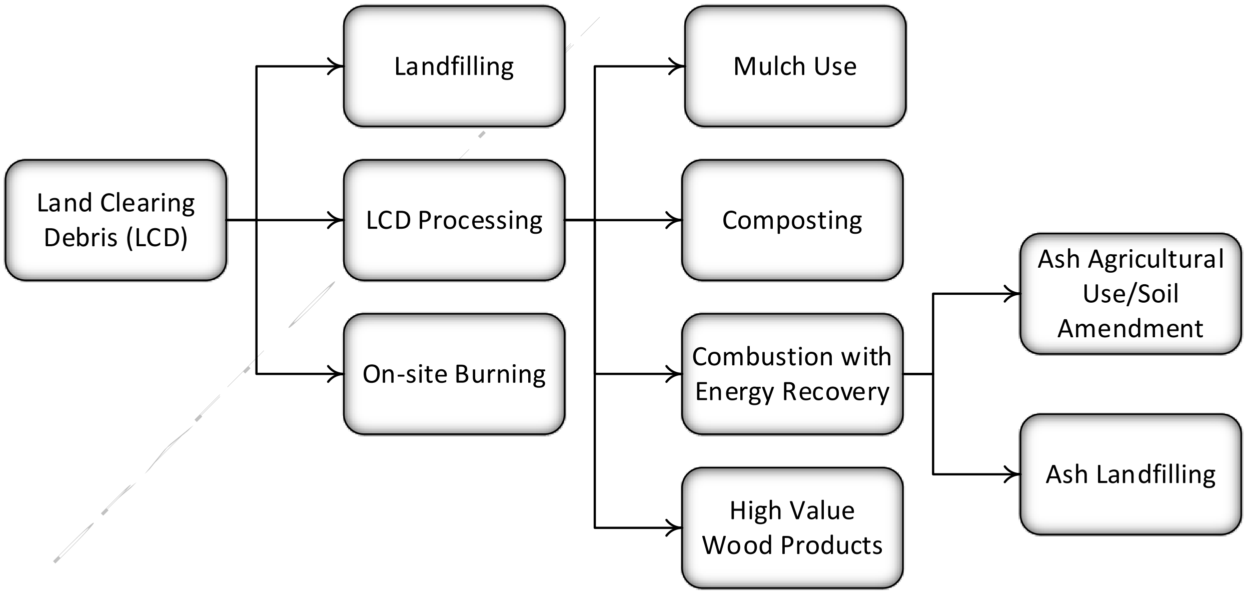

Figure 7-1. Material Flows for LCD and EOL Phase Management

7.2 EOL Management

Based on a survey of LCD contractors, Wiltsee (1998) found that the most common method of LCD EOL management is on-site burning of LCD materials. Air emissions and liquids emission from combustion ash are the primary environmental concerns with burning LCD. LCD, depending on state regulations, may be disposed of in a designated LCD, inert waste, CDD, and MSW landfills that accept LCD. LCD can be used for mulch production, compost production, and biomass fuel. These management options typically require size reducing the debris with a chipper or grinder. Based on a survey of 180 wood collection and processing facilities in 14 counties in Michigan, Nzokou et al. (2011) reported that LCD constituted 61% of the total amount processed by these facilities, and mulch, wood chips, firewood, and industrial fuel were the top four types of recycling products produced at the processing facilities; these materials made up approximately 94% of the recycled products with mulch and chips comprising 42% and 38.6% of all the recycled products, respectively. A nationwide estimate of the quantities of LCD managed with these EOL options is not available.

The emissions associated with composting include gaseous and liquids emissions associated with biological decomposition of LCD. The beneficial uses of compost as a soil amendment include rebuilding the structure of organically-depleted soils, enhancing moisture retention of existent soils, supplanting chemical fertilizer use by supplying depleted nutrients, mitigating pathogens and weed seeds, and remediating/treating contaminated soils (US EPA 1989, Haug 1993).

Chipped LCD can also be used in a boiler for energy recovery. The US EPA (2003) provides a range of heating values for different wood residues (including hogged wood, bark, and chips) on a “wet, as-fired” basis and a dry wood basis as, respectively, 4,500 Btu/lb and 8,000 Btu/lb, with the moisture content of as-fired wood being typically 50% and ranging from 5% to 75% depending on the type of residue and storage techniques.

Table 7-1 lists the processes that should be considered for an LCA of EOL management of LCD. The emissions associated with both energy and materials requirements and process non-energy emissions (e.g., fugitive dust, liquid emissions associated with disposal in a landfill) should be considered for LCA.

Table 7-1. LCD EOL Management Process Descriptions and Considerations for LCA

Process	Description and LCA Considerations
On-site burning	Open burning and air curtain incinerators (ACI) are two approaches associated with burning LCD at its site of generation. The emissions related to the preparation of LCD for combustion, air emissions from combustion and the long-term liquids emission from combustion ash.
Landfill Disposal	Emissions with the materials (e.g., equipment, soil, water) and energy (fuel, electricity) inputs for placement and compaction of LCD in a landfill along with process non-energy emissions (e.g., dust emissions from equipment operation and liquids emission associated with biogeochemical degradation of LCD in a landfill).
LCD Use as Mulch	Emissions associated with grinding and land application, as well as those from leaching due to exposure to precipitation.
LCD Use as Compost	Process energy and non-energy emissions associated with LCD and compost processing and handling as well as long-term liquids emission from land-applied compost.
LCD Combustion for Energy Recovery	Air emission from LCD processing and combustion as well as those associated with management of combustion ash.

7.3 LCI Sources

Peer-reviewed literature, government and private industry publications, and various LCA modeling tools were reviewed to identify available LCI datasets pertaining to LCD EOL management processes. Table 7-2 presents a list of data sources reviewed to develop the LCI presented in this chapter. If LCI data were not available, process metadata and documentation (e.g., included emission categories, background data used to compile the dataset, geographic location, and time period of the data) were reviewed to evaluate the completeness of the dataset. If available, the primary sources of information used to develop the LCI datasets and information were reviewed.

Table 7-2. List of Sources Reviewed for LCI Data

LCI Source	Description
US LCI (2012)	NREL published an LCI database that includes datasets for a wide variety of services and material, component and assembly production processes within the United States.
Cochran (2006)	Cochran (2006) presented diesel energy requirements for C&D wood grinding equipment based on a survey of equipment manufacturers.
Springsteen et al. (2011)	Springsteen et al. (2011) compiled air emissions for open pile burning, wood grinding to produce biomass, biomass transport and combustion in boilers along with energy requirement for biomass production. These data were used to quantify air emission reduction achieved with use of LCD as biomass fuel over open burning for a demonstration project.
WARM	WARM presents data on GHG emissions associated with transport, landfilling (i.e., collection and placement), combustion, and composting of organics.
AP-42 (US EPA 1996, 2002, 2003)	Provides air emissions data for trench air curtain burning, wood chipping, and combusting wood residue in boilers.
MSW-DST	Provides details on composting LCI on processing of yard waste at a composting facility and emissions from land-applying compost.
Ecoinvent	Ecoinvent is an LCI database developed by the Swiss Centre for Life Cycle Inventories which includes specific processes related to the EOL management of numerous individual materials. It includes inventories related to the windrow composting of biogenic waste.
WRATE	Presents LCI data specific to the UK for wood chipping C&D wood materials and composting of yard waste.
EASETECH	Simulates the emissions associated with enclosed windrow composting based on US-specific input data (Komilis and Ham 2004).
Komilis and Ham (2004)	Komilis and Ham (2004) present US LCI data for solid waste composting and provide compost equipment fuel consumption data.
Lutes and Kariher (1997)	The authors present VOC, SVOC, PAH, and criteria air pollutant emission data from burning LCD with and without an air blower.

7.4 LCI Related to On-site Burning

An estimate of the amount of LCD disposed of in the US is not available, but it has been reported that most LCD wood is disposed of at the site of generation. Open burning is one method of LCD disposal; the debris is typically heaped in piles or placed in pits and burned in the absence of emission control devices. As open burning is typically regulated at the state or local level in the US, the emission control requirements and, therefore, emission would depend on location (ERGI 2001).

Combusting LCD using an air curtain incinerator (ACI) is another management approach of LCD at the point of generation. In this case an air blower blows air into the burning debris to enhance combustion, speeding up the combustion process and ideally reducing emissions by achieving more complete combustion. There are two general types of ACIs; a trench ACI is comprised of a mobile air blowing system that is placed along a constructed trench that contains debris and blows air into the trench; another ACI option is a self-contained firebox ACI unit. For the above-ground firebox units, the debris is placed in the firebox, ignited, and the air blower circulates air into the container.

The emissions from burning LCD at its site of generation include those associated with materials and energy input as well as process non-energy emissions released during the preparation of the LCD for combustion and the combustion of LCD. LCI were developed for the on-site burning of LCD through open burning and the use of a firebox ACI. In open burning and ACI scenarios, equipment is needed to arrange the LCD into piles in preparation for burning in the case of open burning or to load the ACI firebox. Cochran (2006) discusses energy consumption for the use of a loader and excavator used to move CDD wood and load a grinder, 11.6 and 0.9 MJ/Mg respectively, at a CDD recycling facility. The fuel consumption for the loader from Cochran (2006) was used as a proxy for moving LCD into piles for open burning (as shown in Table 7-3).

The excavator loading data from Springsteen et al. (2011) were used as a proxy for the fuel consumption for loading LCD into the ACI firebox for combustion (as shown in Table 7-4). Springsteen et al. (2011) documented the average fuel consumption for loading woody biomass into a grinder as 0.79 L of diesel fuel per MT of green material. Operating the blower system of an ACI firebox will also require energy usage. Air Burners's model S-327 ACI specifications were used to approximate the average fuel consumption for a firebox ACI, and was calculated as 1.83 L of diesel fuel per MT of material to be burned (Air Burners 2012). The total diesel consumption for loading and combustion in an ACI was, therefore, estimated to be 2.62 L per MT of LCD.

Springsteen et al. (2011) presented emission factors for nitrogen oxides, particulate matter, non-methane organic compounds (NMOC), carbon monoxide, and methane based on numerous references, including US EPA's AP-42 sections on open burning and wildfires and prescribed burning, laboratory studies, pilot, and full-scale studies on conifer (cone bearing trees) biomass. In the absence of emissions for carbon dioxide and methane for burning LCD in an ACI, the open burning emissions were used as a proxy. Emissions for SVOCs, VOCs, PAHs, and criteria pollutants were collected by Lutes and Kariher (1997) in pilot-scale tests on LCD from two states in the US (Florida and Tennessee). The results from the burning tests (for which greater than half of the data were above the level of detection) were averaged and incorporated into the LCI tables for open burning and ACI air emissions. For data readings below the level of detection, the detection limit was used for the average estimation. US EPA (1996) AP-42 provided estimates of burning wood in a trench ACI. Therefore, sulfur dioxide and nitrogen oxides in the ACI scenario from the US EPA (1996) were used. Table 7-3 and Table 7-4 present the proposed LCI datasets for the open burning and air curtain incineration of LCD, respectively. Because LCD incineration frequently occurs at the site of generation and in the absence of additional information, no LCD transportation is included in the LCI datasets presented.

Table 7-3. Proposed LCI Dataset: Land Clearing Debris, Open Burning

Input Flow	**Source**	**Category**	**Units**	**Amount**
Land clearing debris		Construction and Demolition Debris Management	kg	1
Diesel, combusted in industrial equipment	Cochran (2006)		L	0.0003
Output Flow	**Source**	**Category**	**Units**	**Amount**
Land clearing debris, at open burning		*Construction and Demolition Debris Management*	*kg*	*1*
Carbon dioxide (biogenic)	Springsteen et al. (2011)	Air	g	917
Carbon monoxide	Springsteen et al. (2011)	Air	g	31.50
Methane	Springsteen et al. (2011)	Air	g	1.50
Nitrogen oxides	Springsteen et al. (2011)	Air	g	1.50
Particulates, > 2.5 µm and < 10 µm	Springsteen et al. (2011)	Air	g	2.25
Particulates, < 2.5 µm	Springsteen et al. (2011)	Air	g	7.62
Chloromethane	Lutes and Kariner (1997)	Air	mg	50
1,3-butadiene	Lutes and Kariner (1997)	Air	mg	102
Acetone	Lutes and Kariner (1997)	Air	mg	179
Methylene chloride	Lutes and Kariner (1997)	Air	mg	3.00
Cis-1,2-dichloroethene	Lutes and Kariner (1997)	Air	mg	20.5
2-butanone	Lutes and Kariner (1997)	Air	mg	33.5
Ethyl acetate	Lutes and Kariner (1997)	Air	mg	33.5
Benzene	Lutes and Kariner (1997)	Air	mg	265
Octane	Lutes and Kariner (1997)	Air	mg	5.50
Toluene	Lutes and Kariner (1997)	Air	mg	150
Ethyl benzene	Lutes and Kariner (1997)	Air	mg	24
m,p-Xylene	Lutes and Kariner (1997)	Air	mg	55.8
o-Xylene	Lutes and Kariner (1997)	Air	mg	15.0
Styrene	Lutes and Kariner (1997)	Air	mg	50.8
Pinene	Lutes and Kariner (1997)	Air	mg	48.8
4-ethyltoluene	Lutes and Kariner (1997)	Air	mg	17.3
1,3,5-trimethylbenzene	Lutes and Kariner (1997)	Air	mg	3.50
1,2,4-trimethylbenzene	Lutes and Kariner (1997)	Air	mg	11.8
Limonene	Lutes and Kariner (1997)	Air	mg	46.8
Undecane	Lutes and Kariner (1997)	Air	mg	3.50
Naphthalene	Lutes and Kariner (1997)	Air	mg	53.5

Table 7-4. Land Clearing Debris, at Air Curtain Incineration

Input Flow	**Source**	**Category**	**Units**	**Amount**
Land clearing debris		Construction and Demolition Debris Management	kg	1
Diesel, combusted in industrial equipment	Springsteen et al. (2011) and Air Burners (2012)		L	0.00262
Output Flow	**Source**	**Category**	**Units**	**Amount (wet basis)**
Land clearing debris, at air curtain incineration		*Construction and Demolition Debris Management*	*kg*	*1*
Carbon dioxide	Springsteen et al. (2011)	Air	g	917
Carbon monoxide	Lutes and Kariner (1997)	Air	mg	12.00
Methane	Springsteen et al. (2011)	Air	g	1.5
Nitrogen oxides	US EPA (1996)	Air	g	2.00
Particulates, > 2.5 µm and	Springsteen et al. (2011)	Air	g	0.330
Particulates, < 2.5 µm	Springsteen et al. (2011)	Air	g	10.2
Sulfur dioxide	US EPA (1996)	Air	g	0.05
Chloromethane	Lutes and Kariner (1997)	Air	mg	4.50
1,3-butadiene	Lutes and Kariner (1997)	Air	mg	138
Acetone	Lutes and Kariner (1997)	Air	mg	152
Cis-1,2-dichloroethene	Lutes and Kariner (1997)	Air	mg	20.5
2-butanone	Lutes and Kariner (1997)	Air	mg	24.5
Ethyl acetate	Lutes and Kariner (1997)	Air	mg	24.5
Benzene	Lutes and Kariner (1997)	Air	mg	272
Octane	Lutes and Kariner (1997)	Air	mg	3.00
Toluene	Lutes and Kariner (1997)	Air	mg	189
Ethyl benzene	Lutes and Kariner (1997)	Air	mg	31.0
M,p-xylene	Lutes and Kariner (1997)	Air	mg	119
O-xylene	Lutes and Kariner (1997)	Air	mg	18.0
Styrene	Lutes and Kariner (1997)	Air	mg	72.5
Pinene	Lutes and Kariner (1997)	Air	mg	102
4-ethyltoluene	Lutes and Kariner (1997)	Air	mg	39.0
1,3,5-trimethylbenzene	Lutes and Kariner (1997)	Air	mg	4.50
1,2,4-trimethylbenzene	Lutes and Kariner (1997)	Air	mg	20.0
Limonene	Lutes and Kariner (1997)	Air	mg	71.5
Benzyl chloride	Lutes and Kariner (1997)	Air	mg	2.50
Undecane	Lutes and Kariner (1997)	Air	mg	4.00
Dodecane	Lutes and Kariner (1997)	Air	mg	3.00
Naphthalene	Lutes and Kariner (1997)	Air	mg	47.5

7.5 LCI Related to Landfill Disposal

Emissions associated with LCD disposal in a landfill include air emissions from equipment used for placing LCD in the landfill, emissions associated with landfill construction and operation, and liquids and gaseous emissions from material decomposition in the landfill. The potential for leachate and landfill gas release to the environment depends on the biogeochemical environment of the landfill and the environmental controls, as discussed in Chapter 2. There are various sources that provide landfilling emission factors related to equipment usage and landfill construction and operation; however, none are specific to LCD landfilling. Generalized landfill construction and operations LCI data are also presented in Chapter 2.

The primary constituent in LCD is wood, and as discussed in Chapter 6, due to its organic nature, the decay of wood wastes in an anaerobic (i.e., oxygen poor) environment produces methane, which may be collected by a GCCS and converted to biogenic carbon dioxide via flaring or energy conversion technology, if the landfill has a GCCS present. LCD exposure to precipitation or other liquids (e.g., landfill leachate) is also expected to result in chemical leaching and the emissions are expected to depend on biogeochemical environment (e.g., MSW landfill, CDD landfill). Details on the gaseous emissions produced from decomposition and leachable emissions of untreated wood are presented in Chapter 6.

The emissions from wood decomposition and leaching, energy consumption data from landfill operations, estimated cover soil demand, and an assumed transport distance were used to develop an LCI process dataset for the disposal of LCD at an unlined CDD or inert debris landfill, as presented in Table 7-5. The gaseous and liquid emissions were developed based on the untreated wood waste LCI developed in Chapter 6, but were adjusted for the greater moisture content of LCD. Moisture content of wood used by Jang (2000), Townsend et al. (1999) was assumed to be 10% and a moisture content of 50% was used for LCD; the methane and carbon dioxide emission for wood products disposal in landfills (presented in Chapter 6) are based on a moisture content of 10%. Additional information on how methane and carbon dioxide emissions were estimated for the landfill disposal of LCD can be found in Section 2.5.10.8. The energy use and the associated emissions from landfill operation (e.g., waste placement, compaction) include diesel use in heavy equipment and electricity use in landfill buildings (e.g., administrative buildings, workshop); calculations detailing these emissions and the method of estimating the quantity of cover soil use are also included in Chapter 2 of this report. Diesel consumption from landfill operations and electricity consumption from landfill administrative offices and workshop areas were estimated from Ecobalance (1999) and IWCS (2014b), respectively. In the absence of nation-wide average transport data, it was assumed that LCD would be transported 20 km for landfill disposal. For the purpose of estimating cover soil requirements for the disposal of LCD at an unlined CDD landfill, the density of LCD was estimated from the bulk density of unprocessed forest product fuel wood, as provided Angus-Hankin et al. (1995). It should be noted that inert debris or LCD landfills may not have a cover soil requirement.

Table 7-5. Proposed LCI Dataset: Land Clearing Debris, at Unlined CDD Landfill

Input Flow	Source	Category	Units	Amount
Land clearing debris		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations		Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source		Construction and Demolition Debris Management	kg	0.141

Output Flow	Source	Category	Units	Amount (wet basis)
Land clearing debris, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Methane	US EPA (2012)	Air/Unspecified	kg	0.036
Carbon Dioxide	US EPA (2012)	Air/Unspecified	kg	0.12
Chloride	Jang (2000)	Water/groundwater	mg	41.1
Calcium	Jang (2000)	Water/groundwater	mg	13.3
COD	Jang (2000)	Water/groundwater	mg	1330
Potassium	Jang (2000)	Water/groundwater	mg	23.3
Manganese	Jang (2000)	Water/groundwater	mg	1.22
Magnesium	Townsend et al. (1999)	Water/groundwater	mg	16.1
Carbonate	Townsend et al. (1999)	Water/groundwater	mg	3.39
Sodium	Townsend et al. (1999)	Water/groundwater	mg	4.17

7.6 LCI Related to Recycling

7.6.1 LCD Used as Mulch

LCD recycling generally involves processing (i.e., chipping/grinding) prior to use as boiler fuel, mulch, or other end uses and can occur at the site of LCD generation, mobile equipment, or LCD can be transported to a larger processing facility. The LCI information provided in this section incorporates the transportation of LCD to a large processing facility to produce mulch. Processing of LCD in preparation for mulch production typically involves loading and operating a grinder. Horizontal grinders or tub grinders can be used for grinding vegetative debris. Horizontal grinders are better equipped to handle debris such as tall trees that may be pre-organized prior to being fed into the grinder. Tub grinders, although they can process materials wider in diameter such as tree stumps, root balls, and brushy debris, require long trees to be cut to fit into the tub of the grinder (ESEI 2014). Land clearing applications often involve removing and processing large trees; therefore, a horizontal grinder was the grinding equipment used in the LCD processing LCI presented in Table 7-6.

The emissions from processing LCD include those associated with materials and energy (e.g., transportation and equipment fuel) input as well as process non-energy emission released during grinding/chipping and storing the processed materials. Springsteen et al. (2011) documented the average fuel consumption for grinding woody biomass. The biomass was generated from a prescribed tree thinning in California and included only non-merchantable forest debris. The consumption estimates provided by Springsteen et al. (2011) for loading debris into the grinding with an excavator and grinding the material with a horizontal grinder were 0.79 and 2.92 L of diesel fuel per MT of green material; other studies have reported similar grinder fuel usage (Jones et al. 2010 and Pan et al. 2008).

The process non-energy-related emissions from LCD grinding include particulate matter emission and liquid emission from wood/wood chip stockpiles. AP-42 presents air emission factors for a log chipping operation as a part of an MDF manufacturing operation. These data can be used as a proxy for an LCD wood grinding operation until measurements from operating facilities become available. However, it appears that unlike LCD/CDD processing facilities, engineering controls such as a cyclone and/or fabric filter collection are implemented to control particulate matter emission from chipping operations at MDF manufacturing facilities. The use of log chipping air emission as a proxy would, therefore, underestimate particulate matter emission from an LCD wood processing facility.

As the wood decomposition in this scenario would occur under aerobic conditions, gas emission from land application of mulch were estimated by assuming that 100% of carbon content will decompose to produce carbon dioxide. Using biogenic carbon content of branches (published by Barlaz 1998) as a proxy for wood products, approximately 1.63 kg of carbon dioxide (biogenic) would be produced from aerobic decomposition of 1 kg of wood products; the estimate is based on 0.494 g of carbon content as C per dry kg of wood product and 0.9 kg of dry wood per kg of wet wood product. The liquids emissions from the land application of mulch are expected to be the same as those from wood disposal in CDD materials landfill as leaching is primarily influenced by precipitation. Moisture content of wood used by Jang (2000), Townsend et al. (1999) was assumed to be 10% and a moisture content of 50% was used for LCD; the methane and carbon dioxide emission for wood products disposal in landfills (presented in Chapter 6) are based on a moisture content of 10%.

It was assumed that the LCD wood would be transported 20 km from the job site to the processing facility and the mulch would be transported 20 km; NREL (2013) provides general transport process LCI and are further discussed in Chapter 2. Table 7-6 presents proposed LCI for LCD wood grinding to produce mulch and land application of mulch.

Table 7-6. Proposed LCI Dataset: Ground LCD, Processed and Applied as Mulch

Input Flow	**Source**	**Category**	**Unit**	**Amount**
Land clearing debris		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
Diesel, combusted in industrial equipment	Springsteen et al. (2011)	Flows	L	0.00371
Output Flow	**Source**	**Category**	**Unit**	**Amount (wet basis)**
Ground LCD, processed and applied as mulch		*Construction and Demolition Debris Management*	*kg*	*1*
THC as carbon	US EPA (2002)	Air/Unspecified	kg	1.03E-06
VOC as propane	US EPA (2002)	Air/Unspecified	kg	1.25E-06
Methanol	US EPA (2002)	Air/Unspecified	kg	2.50E-07
Carbon Dioxide	US EPA (2012)	Air/Unspecified	kg	0.906
Chloride	Jang (2000)	Water/groundwater	mg	41.1
Calcium	Jang (2000)	Water/groundwater	mg	13.3
COD	Jang (2000)	Water/groundwater	mg	1330
Potassium	Jang (2000)	Water/groundwater	mg	23.3
Manganese	Jang (2000)	Water/groundwater	mg	1.22
Magnesium	Townsend et al. (1999)	Water/groundwater	mg	16.1
Carbonate	Townsend et al. (1999)	Water/groundwater	mg	3.39
Sodium	Townsend et al. (1999)	Water/groundwater	mg	4.17

7.6.2 LCD Used as Compost

Since windrows are the most common method by which composting occurs in the US, windrow composting was the composting scenario evaluated for LCD. The emissions resulting from composting LCD are those emitted to air from processing the LCD, the gases released during the decomposition of the organic fraction of LCD, and emissions from leaching of chemicals during land application. LCD emissions during processing will include those energy consumption and non-fuel emissions from grinding (as was previously discussed within the mulch production section), those from actively managing the compost, and from post-composting screening. Since windrow composting is the method evaluated in this report, windrow turners or other comparable equipment (e.g., front-end loader) are assumed to be used to turn the windrows on a regular basis during the active composting phase. Once the compost has undergone a period of curing following active composting, the final compost product is screened to remove any large pieces which can undergo a second round of grinding and decomposition, or the screening overs can be disposed of. Other than grinding, it was found there were no processing emissions specific to the composting of LCD; this was expected due to the known scarcity of LCD tracking.

Although the LCI data presented in Komilis and Ham (2004) is not specific to an LCD composting scenario, the fuel consumption of composting MSW and yard waste used in the study are used as a proxy for LCD. The data for estimated diesel consumption for a front-end loader and a windrow turner, both of which could be used to turn windrows, were estimated to be respectively 0.40 and 0.90 L/MT of MSW [US EPA (1991) as cited by Komilis and Ham 2004)]. The electrical requirement for a trommel screen, based on the original data from Diaz et al. (1982) (as cited in Komilis and Ham 2004), to screen out post-composted material was estimated to be 0.8 kWh/MT feedstock material. A summary of optional compost processing fuel consumption data is provided in Table 7-7 in units of per kg "LCD."

Table 7-7. Composting Equipment Energy Consumption per Kilogram of LCD

Equipment	Source	Unit	Amount
Diesel, LCD loading and grinding	Springsteen et al. (2011)	L	0.00371
Diesel, windrow turner	US EPA (1991)	L	0.0009
Diesel, windrow turning with front-end loader	US EPA (1991)	L	0.0004
Electricity, trommel screen	Diaz et al. (1982)	kWh	0.0008

A lack of available material-specific emissions data for LCD composting was observed for gaseous and liquid emissions. LCI data for composting LCD in the absence of other more nitrogen-rich waste organic materials (e.g., yard waste and food waste) were not identified. Wood wastes, such as those in LCD [estimated to be comprised of approximately 97% non-foliage woody material (PGEC 1997)], are high in carbon and thus desirable and commonly used for combination with high nitrogen wastes (e.g., food scraps) as a bulking agent and to create a more nutrient-balanced compost. Since LCD is comprised mostly of wood, it is expected that the air and liquid emissions released from the composting LCD will be proportionally similar to those from the composting of wood. There are multiple data resources that provide gaseous emission estimates for composting; however, these resources are not specific to wood alone, and there is no way to disaggregate the contribution of wood/LCD to the gaseous emissions data from other organic materials (Boldrin et al. 2009, CAR 2010, and Komilis and Ham 2004). Therefore, LCI gas emissions were assumed to be the same as those released from land applying mulch (Table 7-6).

Since compost is generally land applied, similarly to mulching LCD, similar concerns with leaching from the wood from rainfall are implied. In contrast to CDD wood, LCD is not expected to contain potentially harmful wood preservation chemicals. Therefore, liquids emissions from land-applying wood mulch are recommended to be used as a proxy for liquids emissions from LCD compost (Table 7-6).

No LCA modeling tools were identified that included an LCD composting dataset or module. Other organic materials such as yard waste or food waste were generally designated waste materials that can be assessed for use in composting in the following LCA modeling tools and sources: WARM, MSW-DST, Ecoinvent, EASETECH, and WRATE.

7.6.3 LCD Combusted as Boiler Fuel

The two major emissions resulting from the combustion of LCD are those emitted to air and those emitted as a result of leaching of the ash that remains following combustion. Similar to waste wood, LCD may be beneficially used as a boiler fuel for energy recovery, where US EPA (2003) estimates an energy content of approximately 18.6 MJ per kilogram dry wood material. LCD is mainly comprised of woody material; PGEC (1997) reports that approximately 97% of LCD (by mass) is comprised of non-foliage woody material; therefore, it is expected that the air emissions released from the combustion of LCD will be proportionally similar to those from the combustion of dimensional wood. However, emissions from the combustion of LCD will need to be adjusted for moisture content, which is estimated at 50% (wet basis) as presented in several sources for woody biomass and “wet” wood materials used as feedstock for boilers (Maker 1994, US EPA 2003, Tumuluru et al. 2011).

An additional difference associated with the combustion of LCD is the ash fraction of the feedstock material. The ash content of woody biomass waste as delivered to a power generation facility in California was found as 2% (dry basis) (Springsteen et al. 2011), which appears to be consistent with the ash content of tree species presented in Jenkins et al. (1998). A review of government and peer-reviewed literature did not provide information on leaching emissions from the landfilling or land application of ash from LCD combustion; however, there have been leaching studies performed on wood ash [e.g., Tolaymat (2003) performed leaching tests on ash from combusted wood and tires]. In the absence of specific data for LCD, it is proposed that the untreated wood ash leaching dataset provided in the wood product chapter be used to simulate emissions associated with disposal of LCD ash in landfills.

The NREL/US EPA LCI database includes a “Combustion, wet wood residue, AP-42” process developed from US EPA (2003) AP-42 emission factors which simulates wet wood residue combustion in a boiler. It is recommended that this process dataset be used to model the emissions and energy recovery associated with the combustion of LCD. However, it is recommended that the process dataset be updated to include the mass flow input of processed LCD as described elsewhere in this chapter and to include the output mass flow of ash resulting from the combustion process. This modification would allow the inclusion of this combustion process dataset in a product system that models the overall emissions associated with this EOL management of LCD from clearing operations to final deposition of ash.

Additionally, the “Combustion, wet wood residue, AP-42” process does not include processing of the wood prior to combustion. Woody biomass must first be processed (by chipping or grinding) prior to being combusted in a boiler, therefore dataset Table 7-8 (adapted from Table 7-6, LCD mulching) is proposed for estimating the emissions from transporting and grinding LCD prior to combustion.

Table 7-8. Proposed LCI Dataset: Ground LCD, at Processing Facility

Input Flow	Source	Category	Unit	Amount
Land clearing debris		Construction and Demolition Debris Management	kg	1
Transport, single unit truck, short-haul, diesel powered - US	Assumed		t*km	0.001*20
Diesel, combusted in industrial equipment	Springsteen et al. (2011)	Flows	L	0.00371
Output Flow	**Source**	**Category**	**Unit**	**Amount (wet basis)**
Ground LCD, at processing		*Construction and Demolition Debris Management*	*kg*	*1*
THC as carbon	US EPA (2002)	Air/Unspecified	kg	1.03E-06
VOC as propane	US EPA (2002)	Air/Unspecified	kg	1.25E-06
Methanol	US EPA (2002)	Air/Unspecified	kg	2.50E-07

7.7 Data Gap analysis and Opportunities for Additional LCI Data

Table 7-9 summarizes the type of US-based LCI identified from reviewed compilations of LCD EOL management sources. Most of the data identified are recognized as being partial as these provide only energy consumption data or emissions from one aspect (e.g., air, water, materials) of EOL management for LCD. Overall, limited EOL-specific LCI are available for LCD, which is likely a result of limited material management tracking of LCD. Based on a review of the available information, the following data gaps were identified for compilation of a more comprehensive LCI dataset for LCD EOL management:

1. **Data pertaining to LCD generation and disposal estimates and EOL management practices.** Scarce information is available on the quantities of LCD that are managed in the US, which is likely due to variable or absent tracking systems at the state and local level; for example, some states consider LCD as CDD whereas other do not. Thus, evaluating LCD management in the EOL phase is difficult due to a lack of quantitative data available for LCD. As these data are of interest to multiple governmental agencies (e.g., US EPA, USDA, state environmental agencies), an opportunity for collaborative research exist to quantify current practices of LCD management in the EOL phase.
2. **Long-term leachable emissions from LCD products placed in a landfill.** As described earlier, the liquid emission presented in this study are based on SPLP and TCLP tests on untreated wood, which simulates leaching from disposal in inert debris landfill (or land-application), and MSW landfill, respectively. The leaching test data of untreated dimensional lumber was used as a proxy for liquids emission estimate for LCD due to lack of data. Although woody material is the primary constituent of LCD, other LCD constituents such as leaves, roots, stems, bark may impact liquid emissions. Moreover, the batch leaching data used for estimating liquid emissions correspond to L:S ratio of 20 and are, therefore, not representative of complete liquid emission. Furthermore, the standardized leaching tests simulate leaching associated with physical and chemical mechanisms and do not simulate leaching associated with biological decomposition of wood due to the short duration of these tests (18 hours). Future research should consider assessment of leaching over a greater L:S ratio and those associated with biological decomposition, leaching should also consider observing for the presence of pesticides or herbicides that may have historically been used to control insects and vegetation where LCD is generated.
3. **Long-term gaseous emission from LCD biodegradation in landfill**. The data reported for branches were used as a proxy for estimating gaseous emission from anaerobic biodegradation of

LCD disposed of in landfills due to lack of LCD-specific data. Moreover, the emissions of only two compounds (methane and carbon dioxide) are included in the proposed LCI for landfill disposal of wood/LCI products. Future research should consider quantifying a larger suite of gaseous emission from LCD to assess the impact of large amounts of LCD disposed of in landfills collecting primarily just LCD materials.

4. **Materials and energy input and emission from LCD processing.** Emissions from logs chipping operation for medium density fiberboard (MDF) were used as a proxy for emissions from LCD grinding. Particulate matter emission, from log grinding for MDF manufacturing, however, were not detected likely due to air pollution control practices for MDF manufacturing operations. The proposed LCI for LCD processing does not include particulate matter emission and liquids emission from short-term LCD stockpile to water due to the lack of these data. Future research should consider collecting and compiling these data. Some of the data for processing (e.g., consumables, fuel and electricity usage, water consumption, material throughput) tracked by LCD processing facility owners from a financial accounting perspective can be readily used for developing more comprehensive LCI for LCD processing.
5. **LCD composting.** There is a large body of literature available for composting organics (e.g., yard waste, food waste). The data pertaining to LCD composting, however, are lacking; yard waste data were not used as proxy as yard waste composition is significantly different from LCD. Future research should consider quantifying air and liquid emissions from LCD composting operations.
6. **LCD combustion (including onsite LCD burning) ash.** Numerous studies characterized wood residue ash to assess its benefits as soil amendment (e.g., as lime substitute). Leaching data (SPLP), however, are inadequate to assess nutrients released from land-application of LCD. Future research should consider quantifying leaching emission from onsite burning and combustion for energy projects for land application and various disposal scenarios.
7. **ACI emissions data for onsite burning of LCD.** Research has been conducted on open burning of LCD and related materials; there have been some studies conducted with ACIs burning woody materials and one instance of burning LCD (Lutes and Kariher 2004) from which data were collected which could be developed into emission factors. Although it has been debated that ACIs likely improve emissions compared to emissions from open burning, little data with respect to LCD, confirming the effectiveness of ACIs burning LCD. Future research should consider collection and compilation of these data since burning LCD at the site of LCD generation is still presumed to be the most commonly used EOL management strategy.

Table 7-9. Overview of LCD Data Available

Process	**AP-42**	**US EPA/ NREL**	**Springsteen et al. (2011)**	**Cochran (2006)**	**Komilis and Ham (2004)**	**Lutes and Kariher (1997)**
Onsite Disposal by Burning	P					P
Transport		X				
Landfill Leachate						
Landfill Gas Emissions						
Mulch Processing and Use	P		P	P		
Compost Processing and Use					P	
Combustion of LCD for Energy	P	X	P			
Ash landfilling						

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8 Portland Cement Concrete

8.1 Introduction

PCC is a composite material formed from fine aggregates (i.e., sand), coarse aggregates (e.g., gravel, crushed stone), binder (Portland cement), water, and stabilizers. By volume, aggregates, cement, and water represent 60-70%, 10-15%, and 15-20% of the concrete mix, respectively (PCA 2014a). The remaining volume of concrete is entrained air and any additional stabilizers or other amendments added to enhance desired properties of the concrete. Concrete is widely used in the construction industry due to its versatility, strength, and cost – (assuming a density of 150 pounds per cubic foot) nearly 480 MMT of ready-mixed concrete is used annually in the US (PCA 2014b). Ready-mix concrete is the most commonly used concrete type and accounts for about three-fourths of all concrete used annually (PCA 2014b).

Pavements, bridges, and various components of airports and buildings that have been constructed from concrete may be rehabilitated or demolished and reconstructed due to wearing or damage that has occurred over time. Concrete may be removed by different techniques (e.g., blasting, crushing, cutting, impacting, milling, and presplitting), which are determined based on factors such as cost, project duration, the quality of the concrete, the potential for recycling, transport distances, and accessibility (Lee et al. 2002, Lechemi et al. 2007 and Woodson 2009).

Once removed, reclaimed PCC may be recycled or disposed of in a landfill. The concrete is typically processed (e.g. crushing, sorting, metal removal) prior to use in a recycling application. Figure 8-1 identifies the flow of materials and processes that should be considered for conducting an LCA of concrete EOL management. Most commonly, recovered concrete is recycled as aggregates (referred to here as *recycled concrete aggregate [RCA]*) in road base, for new concrete mix, or for asphalt pavement mix production. Closed-loop recycling of concrete, where RCA replaces both primary aggregate and cement, is not a common practice at present; the US EPA (2012) indicated lack of data for developing emission factors for closed-loop recycling of concrete.

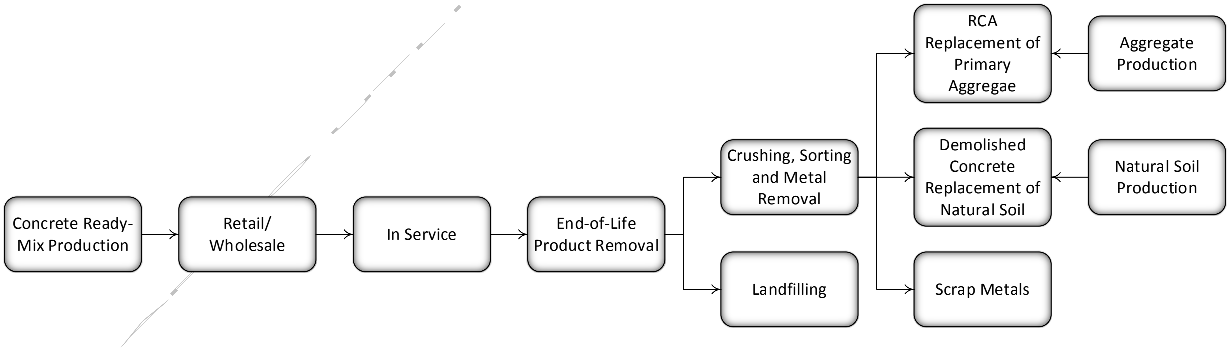

Figure 8-1. Materials Flow for Concrete EOL Phase Management

8.2 EOL Management

As presented in Figure 8-1, there are three primary EOL management pathways for concrete: use as RCA, use as general fill, and landfill disposal. CDRA (2014) reports that approximately 127 MMT of concrete are recycled annually; however, the basis of this estimate of recycled concrete is not well documented. Concrete (with and without rebar, painted and unpainted concrete) represents approximately 10.8 to 15.2% (by mass) of CDD materials received at CDD landfills (CCG 2006, CDM 2009, and RWB et al. 2010). Based on US EPA's estimate of total CDD landfilled (as presented in Chapter 2 of this report), approximately 9.5 to 13 MMT of concrete was disposed of in landfills in 2011. However, based on information from CDRA (2014), Turley (2002) and Wilburn and Goonan (1998), it appears that the total amount of concrete recovered for EOL management in the US on an annual basis may range between 212 to 254 MMT, though the primary sources of data used to develop the estimates provided in these documents are not clearly provided.

An earlier dated estimate, as derived from various USGS and industry sources, presented by Wilburn and Goonan (1998) reported that approximately 50% of cement concrete debris generated in the US is landfilled and the rest is recycled. Of the recycled amount, 86% is used as road base, 8% is used in asphaltic concrete (i.e., a total of 94% is used as RCA), and 6% is used as general fill (Wilburn and Goonan 1998). However, these end-use projections are based on the USGS estimate of a total of 14.5 MMT generated in the US, which is likely a substantial underestimation of the actual amount of concrete debris generated in the US.

Deal (1997) (as cited in Kelly 1998 and USGS 2000) also presented recycled concrete uses; a copy of Deal (1997) was not available for more details of the projections presented. Figure 8-2 presents the distribution of concrete debris used in different applications in the US based on US EPA's estimate of concrete landfilled, CDRA's estimate of concrete recycled, and the distribution of concrete uses reported by Deal (1997). The use as aggregate for road base is the most common management option for RCA (FHWA 2004, CTCA 2012). Use of RCA in asphalt mixes is another desirable option as it can improve the stability and surface friction of the pavement apart from offsetting production of primary aggregate (Snyder and Rodden n.d.). However, the use of RCA in asphalt mixes can increase the need for greater asphalt content in the paving mix due to RCA's absorptive properties (Snyder and Rodden n.d.).

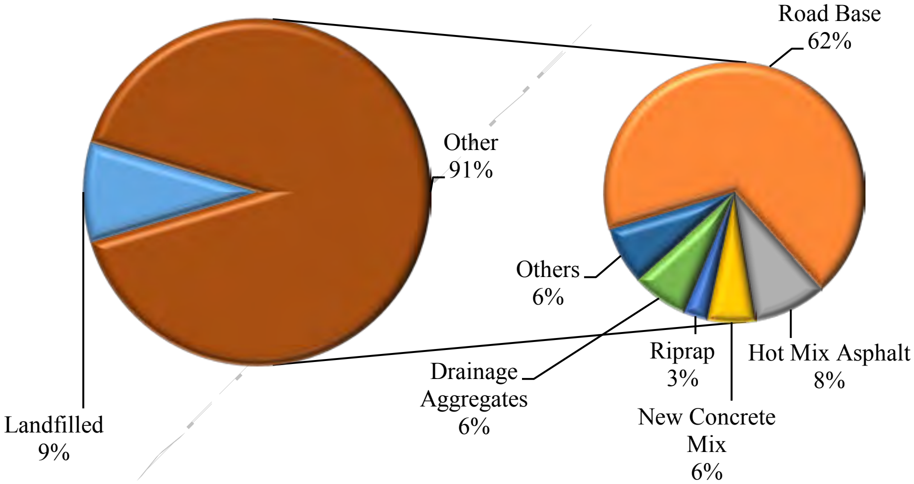

Figure 8-2. Distribution of Recycled Concrete Uses in the US (Deal 1997)

Decreased workability as a result of its angular structure, reduced durability due to potential alkali-silica reaction, and reduced compressive strength when substituted for fine aggregates or substituted for more than 30% of coarse aggregates are some of the challenges of recycling RCA into new concrete mixes (Hansen 1986, Li and Gress 2006, McIntyre et al. 2009 and Hiller et al. 2011). In a survey of state concrete recycling practices, 10 out of 30 responding states allowed the use of crushed concrete for road surface course (CTCA 2012). However, only two Alabama and Texas reported this use as a common practice.

Recycled concrete can also be used as riprap. FHWA (2004) reported that most states allow processed recycled concrete to be used as riprap for erosion control, as long as steel reinforcement has been removed prior to use. Demolished concrete can also be used in fill applications (e.g., embankments) as a substitute for natural soil. The "drainage aggregates" includes uses such as drainage fields and pipe bedding. The "Other" category includes use as railroad ballast and landscaping rock.

Table 8-1 lists the processes that should be considered for an LCA of EOL management of concrete. The emissions associated with energy and materials requirements and process non-energy emissions (e.g., fugitive dust, liquid emissions associated with disposal of concrete in a landfill) were taken into account in compiling the different LCI datasets.

Table 8-1. Concrete EOL Management Process Descriptions

Process	Description
Concrete Removal/Demolition	Concrete may be removed by a variety of processes at the end of serviceable life and the mechanism of removal will depend on project needs and constraints and intended end-use of recovered concrete.
Transport	The emissions associated with the transport of demolished concrete to a recycling facility or a landfill, primary aggregate and RCA to end users should be considered for LCA.
Landfill Disposal	The materials (e.g., equipment, soil, water) and energy (fuel, electricity) inputs for placing and compacting discarded concrete in a CDD landfill along with process non-energy emissions (e.g., dust emissions from equipment operation and liquids emission associated with physiochemical degradation of concrete in a landfill) should be included in LCA.
Concrete Processing	Concrete processing includes crushing, sorting, and removing metal. The extent of concrete processing depends on the end use of the processed material. Concrete can be processed on-site or off-site using mobile or stationary equipment.
RCA Replacement of Primary Aggregates	RCA may be used as primary aggregate substitute in a variety of applications, including road base construction, asphalt or concrete mix production, or as riprap. This use of RCA precludes the production of an equivalent amount of primary aggregate.
Concrete Replacement of Natural Soil	Concrete may be used as a soil substitute in fill applications. When used as a fill material, the concrete will likely not need the processing and sorting requirements necessary for using concrete as aggregate. This use of demolished concrete precludes the production of natural soil.

8.3 LCI Sources

Peer-reviewed literature, government and private industry publications, and various LCA modeling tools were reviewed to identify available LCI datasets pertaining to concrete EOL management processes. Table 8-2 lists sources reviewed to develop the LCI presented in this chapter. If LCI data were not available, process metadata and documentation (e.g., included emission categories, background data used to compile the dataset, geographic location and time period of the data) were reviewed to evaluate the completeness of the dataset. If available, the primary sources of information used to develop the LCI datasets and information were reviewed. The USLCI, EPA, and GaBi database also present LCI for various processes primarily related to cement and aggregate production.

Table 8-2. List of Sources Reviewed for LCI Data

LCI Source	Description
Wilburn and Goonan (1998)	The authors provide energy requirements associated with crushing/sorting stone, sand and gravel, and concrete. These data were taken from the Portland Cement Association and an energy audit of a recycling facility in Denver, Colorado.
Stripple (2001)	H. Stripple, on behalf of the IVL Swedish Environmental Research Institute, presents a LCA of Road for the Swedish National Road Administration. The document provides LCI for a wide variety of road construction, pavement production, maintenance, and demolition activities (e.g., land clearing activities for road placement, installation of signs). Emissions are based on information gathered from 1990-1994 for numerous road manufacturing and upkeep processes taken from a variety of industry and heavy equipment manufacturer sources.
US EPA (2012)	EPA’s Waste Reduction Model presents data on GHG emissions associated with transporting, recycling, and landfilling (i.e., collection and placement) concrete.
Ecoinvent (2014)	Ecoinvent is an LCI database developed by the Swiss Centre for Life Cycle Inventories, which includes specific processes related to the EOL management (demolition, processing, and disposal) of numerous individual materials, including waste concrete.
Cochran (2006)	Cochran (2006) presented diesel energy requirements for concrete processing equipment based on a survey of equipment manufacturers.
Jang (2000)	Jang (2000) presents batch and column leaching tests data for various CDD materials, including concrete.
McIntyre et al. (2009)	McIntyre et al. (2009) presents energy and GHG emissions savings estimates for RCA production and substitution for primary aggregates in nonstructural applications.
NIST (2007) BEES	The National Institute of Standards and Technology Building for Environmental and Economic Sustainability model allows an economic and environmental impact comparison among various building materials, including concrete.

8.4 LCI Related to Removal/Demolition

Concrete demolition generally includes breaking the concrete into manageable chunks for ease of handling and transportation. In-place concrete characteristics may be analyzed to assess properties and suitability for use in targeted applications prior to demolition (Hiller et al. 2011). Contaminants such as joint sealant and large portions of asphalt overlay or patch are recommended to be removed prior to concrete demolition, but, depending on the RCA application, small amounts of asphalt contamination are not detrimental (CDRA 2014). The project location and nature (e.g., need to protect the surrounding infrastructure) dictate the concrete removal method and equipment used (Lechemi et al. 2007 and Woodson 2009). Dykins and Epps (1987) and NHI (1998) [as cited in Hiller et al. (2011)] describe two general types of equipment that can be used for breaking up concrete in highway applications to render concrete into sizes acceptable for crushing: impact breakers and resonant breakers. Impact breakers use individual weighted drops to break the concrete and this equipment has greater production rates than resonant breakers (900 to 1,100 m^2/hr compared to 670 m^2/hr), which use a high-frequency, low-amplitude pulse to fracture the concrete. Resonant breakers have the advantage of producing more uniform slabs and causing fewer disturbances to underlying infrastructure (e.g., sewers, utilities).

After the concrete has been fractured and separated from reinforcement, larger pieces of concrete (<24") chunks of concrete can be removed using a backhoe while a front end loader can be used to remove the remaining pieces. The demolished concrete may be processed onsite or transported to an offsite recycling/processing facility to produce RCA or may be transported to a landfill for disposal.

The emissions for demolishing concrete include those associated with demolition equipment manufacture and EOL management, operation and maintenance consumables (e.g., lubricants, air filters, belts), energy (fuel) inputs, and particulate matter releases during demolition. The US Advisory Council on Historic Preservation (1981) [as cited in Cole and Kernan (1996)] provided the energy needed for demolishing a concrete building on a MJ/m^2 basis. However, the energy uses (e.g., transportation, equipment fuel usage) and the calculation methodology of this estimate were not clearly defined. Weiland and Muench (2010) used the US EPA NONROAD 2005 model to estimate air emissions from fuel consumption in equipment used for breaking and loading PCC as part of an LCA for concrete pavement. As NONROAD does not include a pavement breaker, a combination of off-road truck and a crushing/processing equipment was used to estimate emission from a pavement breaker. Although fuel usage (BTU/hr) for this equipment was reported, the concrete demolition rate (MT/hour) was not reported to estimate energy used for demolishing a unit mass of in-place concrete.

Various European and other non-US literature sources have published energy consumption and emissions information associated with demolishing or dismantling concrete structures. The Ecoinvent database includes estimates of diesel fuel consumption for dismantling reinforced (0.0612 MJ/kg) and non-reinforced concrete (0.0437 MJ/kg) based on dismantling practice in Switzerland (with hydraulic diggers) (Doka 2003). Ecoinvent inventories particulate matter (PM) emissions from building construction, demolition, renovation, and highway reconstruction; however, it does not distinguish between PM related to concrete handling and the handling of other mineral construction materials (e.g., bricks, cement, gypsum, plaster). MGE (1997) presented demolition energy estimates for three structural materials (wood, steel, and concrete) for Canada based on factors such as typical equipment used, fuel used, and energy consumption rates. The estimated energy for demolishing a concrete structure (including cutting reinforcing steel) was 0.0681 MJ/kg concrete. Stripple (2001) published energy consumption from milling a concrete road in units of liters of diesel per area of concrete milled and mega joules per area of milled concrete in Sweden. LCI projecting the environmental burdens from concrete structure demolition could not be compiled due to lack of data.

8.5 LCI Related to Disposal

As discussed earlier, based on published literature and US EPA's compilation of state data, it is estimated that approximately 9% of the concrete generated in the US is disposed of in landfills. Emissions associated with concrete disposal in a landfill include air emissions from equipment used for placing concrete in the landfill, emissions associated with landfill construction and operation, and liquid emissions from material decomposition in the landfill. There are various sources which provide landfilling emission factors related to equipment use and landfill construction and operation; however, none are specific to concrete landfilling. Generalized landfill construction and operations LCI data are presented in Chapter 2.

Concrete exposure to precipitation or other liquids (e.g., landfill leachate) is expected to result in contaminants leaching and the emissions are expected to depend on biogeochemical environment (e.g., MSW landfill, CDD landfill). Leachable emissions were estimated using batch leaching tests (SPLP) and leaching column data reported by Jang (2000). Jang (2000) conducted leaching tests on several individual CDD materials, including concrete (size reduced to 1") collected from a concrete recycling facility in Florida to assess leaching of conventional water parameters [e.g., pH, conductance, TDS, and COD], ions and heavy metals. The L:S ratio of the batch tests in Jang (2000) were much greater than the L:S ratio in the column tests (20:1 versus 1.3:1); batch data were used for parameters (calcium, chloride, potassium and sodium) that were measured above the detection limits (because of the greater propensity to leach pollutants). Batch test concentrations were multiplied by the total solution volume and divided by the sample mass to estimate leachability on a per-kilogram-concrete basis.

For parameters that were either below the detection limit in the batch testing experiment or were not measured during the batch test, column test data were used to develop leaching LCI. Leachability of COD and magnesium were calculated from column test data by summing the total mass of pollutant leached and dividing this mass by the mass of the concrete material in the column. Non-purgeable organic compound emission was not estimated as this compound is not included as a flow in US LCI (2012) datasets. Nitrate and sulfate emission data reported by the study were not used for developing the LCI dataset since SPLP extraction fluid contains these anions. TDS data reported by the study were also not included to avoid double-counting emissions, as some of the contaminants listed in Table 8-3 are included in TDS measurement.

Concrete leaching data, energy consumption data from landfill operations, estimated cover soil demand, and an assumed transport distance were used to develop an LCI process dataset for disposing of concrete at an unlined CDD or inert debris landfill, as presented in Table 8-3. The energy use and the associated emissions from landfill operations (e.g., waste placement, compaction) include diesel use in heavy equipment and electricity use in landfill buildings (e.g., administrative buildings, workshop); calculations detailing these input flows are included in Chapter 2 of this report. The bulk density of loose concrete used to estimate cover soil requirements was provided by CCG (2006); additional details on the cover soil estimate are provided in Chapter 2. In the absence of nationwide average transport data, it was assumed that concrete would be transported 20 km for landfill disposal. US LCI (2012) provides general transport process LCI, which are further discussed in Chapter 2. Emissions are provided per kilogram "Concrete, at unlined CDD landfill" flow.

Table 8-3. Proposed LCI Dataset: Concrete, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
Concrete, from demolition		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.0372
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Concrete, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Calcium	Jang (2000)	Water/Groundwater	mg	2860
Chloride	Jang (2000)	Water/Groundwater	mg	74.0
COD, Chemical Oxygen Demand	Jang (2000)	Water/Groundwater	mg	49.4
Magnesium	Jang (2000)	Water/Groundwater	mg	1.44
Potassium	Jang (2000)	Water/Groundwater	mg	240
Sodium	Jang (2000)	Water/Groundwater	mg	52.0

8.6 LCI Related to Recycling

8.6.1 Concrete Processing

The recycling end-use and corresponding specifications and project and end-use locations dictate the processing degree and location. For example, no processing may be needed if concrete is used to replace natural soil fill in a non-load-bearing fill application. RCA production, on the other hand, requires extensive processing. Concrete may be processed on-site using mobile equipment if the processed concrete is intended to be used at the site. It may be transported for off-site processing for future end uses. Concrete commingled with other CDD materials would need to be segregated prior to processing. Additional details on CDD processing facilities are provided in Chapter 2 of this report.

Off-site processing facilities may have a multi-phase crushing operation in which the concrete passes from a primary crusher to a secondary crusher. Several types of crushers can be used in concrete processing: jaw crushers are typically primary crushers, cone crushers are secondary crushers, and impact crushers can be both primary and secondary crushers. While impact crushers have the advantage of removing a greater amount of mortar from concrete aggregates (yielding a better quality aggregate), removing more mortar results in a larger amount of (typically landfilled) fine material (Hiller et al. 2011). Hiller et al. (2011) reported that the overall grading of RCA is difficult to control in a concrete crushing operation and the crushed concrete, typically, is gap-graded (i.e., mostly consists of large and fine aggregate, but generally only has a small amount of mid-size aggregate). The crushed concrete may need to be screened to achieve a desired gradation (e.g., a scalping screen may be used to remove excess dirt and foreign particles or a fine harp deck screen may be used to remove fine material from coarse aggregate). The RCA recovery rate (amount of RCA produced per unit mass of concrete processed) depends on the targeted maximum aggregate size (Hiller et al. 2011). The recovery rate is greater for crushing operations with greater maximum particle size of RCA. For example, a recovery rate of 80% is reported from a crusher set to produce RCA with a maximum particle size of 38 mm, whereas the recovery rate is less than 60% for crushing operations producing RCA with a maximum particle size of 19 mm (Hiller et al. 2011).

Magnetic separators may also be incorporated between crushing processes to remove steel (e.g., mesh, rebar, dowels) from demolished concrete. The fraction of steel recovered from concrete recycling depends on how easily the steel can be separated from the concrete. For example, wire mesh (generally used for reinforced pipes) usually retains a large quantity of concrete and is not recovered [ACPA (1993) (as cited in FWHA 2004)]. NIST 2007 reported 135 lb of steel per cubic yard of concrete as the industry average steel content of concrete. However, no data regarding distribution of reinforced and non-reinforced concrete and steel recovery rates from concrete crushing operations were found.

Facilities may store and process concrete by material source to maintain a homogeneous RCA quality (TRB 2013). The processing facilities may need to temporarily stockpile the incoming concrete debris and RCA depending on demand for the end product as a result of this practice. Liquids emission associated with exposure to natural precipitation has been reported to be an issues associated with concrete/RCA stockpiles. Various departments of transportation and research groups have identified the potential for runoff (i.e., leachate) from unbound RCA to be highly alkaline (reaching up to a pH of around 12) (Snyder 1995, Steffes 1999, Jang 2000, Hiller et al. 2011 and Chen et al. 2012). Water used for controlling particulate matter emission from crushing operation is another source of liquid emission. The emissions from stockpiling should, therefore, be considered for an LCA on recycled concrete. No literature reporting water consumption for dust control was found. Another issue with RCA stockpiling is that the material is susceptible to cementing since RCA may contain some amount of unhydrated cement; therefore, the material should be protected from moisture to prevent agglomeration (Hiller et al. 2011).

The emissions from processing waste concrete include those associated with materials and energy input as well as non-energy emissions released during crushing, screening, and storing the processed materials. MGE (1997) estimated on-site concrete debris processing energy requirement of 5 MJ/MT; processing included stockpiling, preparing the concrete for crushing (size reducing from 380 mm to 200 mm), loading the crusher, and crushing the concrete to 63-mm aggregate. Cochran (2006) used equipment manufacture energy consumption and approximate production rates to estimate the energy required to operate a concrete crushing operation, which includes use of an impact crusher, excavator, and loader. Cochran (2006) estimated that approximately 7 MJ/Mg (total) of energy would be consumed by the equipment; however, this estimate does not include additional energy consumption from ancillary buildings, equipment, or processes. Wilburn and Goonan (1998) reported an energy requirement of 34 MJ for crushing/sorting 1 MT of PCC based on data from the Portland Cement Association and data from a recycling facility in Colorado; these data do not explain what sources are consuming the energy (e.g., buildings, equipment). Wilburn and Goonan (1998) also provide transport energy requirements recycled concrete in terms of joules per kg-km. Wilburn and Goonan (1998) is commonly cited (e.g., within McIntyre et al. 2009, WARM model, Cochran 2006) as a source for concrete processing data.

The energy estimates provided by Wilburn and Goonan (1998) are used to create a generalized energy estimate for the concrete processing LCI presented in Table 8-4. In the absence of additional data, electricity and diesel consumption were each assumed to constitute 50% the total energy requirements. A similar approach was used by the US EPA (2012) in estimating GHG emission factors for demolished concrete processing for WARM; US EPA (2012) also used the energy consumption data reported by Wilburn and Goonan (1998) to estimate the emission factors for WARM.

Concrete crushing and sorting particulate emissions were not identified from US Sources. As discussed in Chapter 2, US EPA AP-42 (1995) does not provide PM emissions for primary and secondary stone crushing operations (the most similar industrial process to concrete crushing that has quantified emissions). Estevez et al. (2008) documents 0.14 g dust per MT of recycled concrete and other air emissions from mobile concrete crushing operations in Spain. Due to lack of data (quantity and quality of liquid emission from concrete/RCA stockpile) leachate/runoff emissions to groundwater/surface water from RCA stockpiling were not included in the concrete processing dataset.

Table 8-4 presents the proposed LCI dataset for concrete processing that occurs at a concrete waste generation site with mobile equipment or at an offsite processing facility. Similar to aggregate crushing and sorting during production, it is expected that particulates would be a source of emissions released from concrete crushing and sorting operations. However, as explained previously, due to a lack of data, fugitive dust emissions released during the crushing process and emissions from the manufacturing, maintenance, and disposal/dismantling of the crushing equipment are not included in Table 8-4. This dataset should not be used if recovered concrete is not processed prior to end use (e.g., application to replace a soil embankment fill). In the absence of national average data on the distance from the demolition site to offsite concrete processing, a distance of 20 km was assumed, US LCI (2012) provides general transport process LCI, which are further discussed in Chapter 2. It should be noted that on-site concrete processing operations would likely result in fewer emissions associated with transport because transport emissions will typically include only those associated with equipment mobilization.

Table 8-4. Proposed LCI Dataset: Crushed Concrete, at Processing Plant

Input Flow	Source	Category	Unit	Amount
Concrete, from demolition		Construction and Demolition Debris Management	kg	1
Diesel, combusted in industrial equipment	Wilburn and Goonan (1998)	Flows	L	0.00044
electricity, at industrial user	Wilburn and Goonan (1998)	Flows	kWh	0.00472
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Crushed concrete, at processing plant		*Construction and Demolition Debris Management*	*kg*	*1*

8.6.2 RCA Use as Aggregate

Recycled concrete is most commonly used to replace primary aggregate in the following applications: road base, subbase, asphalt and cement concrete, and riprap. RCA produced from demolition activities will likely need to be crushed, processed to remove contaminants such as steel, and sorted prior to use in these applications to meet gradation specifications. The primary emissions resulting from the use of RCA as an aggregate material (after the concrete has been processed) include leaching to groundwater/surface water. Granular base (unbound) applications of RCA have been shown to leach calcium carbonate precipitate that can clog drainage pipes (particularly if there is a large amount of fine material in the RCA) and may restrict use in various drainage applications (Gupta 1993, Snyder 1995, Steffes 1999). Although there have been no reported problems with this precipitate for embankment applications, there is the possibility for highly alkaline precipitates to occur (FHWA 2012). The liquid emissions from unbound RCA are expected to be similar to those from an inert landfill. While it would take longer for bound RCA (e.g., use in asphalt pavement, use in new concrete mixes) to leach contaminants, it is expected that over an infinite time horizon, these emissions would ultimately be the same.

The use of RCA in an aggregate-required fill application avoids the need for the production of primary aggregates; emissions associated with primary aggregate production and transport are presented in Chapter 2. The US EPA (2003) background documentation for the WARM model used a distance of 15 miles for the transport distance of recycled aggregate material; however, the justification for this distance was not provided. Again, because the transport distance from the processing facility to the aggregate fill site is unknown, this distance was assumed to be 20 kilometers. Table 8-5 presents the proposed LCI dataset for RCA used to replace primary aggregate.

8.6.3 Demolished Concrete Use as Soil Fill Replacement

The use of demolished concrete to replace natural soil fill (e.g. lake fill, embankment fill) would avoid the emissions resulting from the production and transport of natural soils. Table 8-6 presents the proposed LCI dataset for demolished concrete used to replace natural soil. While leachable emissions are assumed to ultimately be the same whether demolished concrete is used as a substitute for bound aggregate, unbound aggregate, or natural soil, this dataset assumes that the concrete debris used as a fill material does not need to be size reduced or screened. It was assumed that concrete debris will be transported 20 km from the demolition site to the fill site.

Table 8-5. Proposed LCI Dataset: Recycled Concrete Aggregate, Use as Aggregate

Input Flow	Source	Category	Unit	Amount
Processed concrete, at processing plant		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Recycled concrete aggregate, use as aggregate		*Construction and Demolition Debris Management*	*kg*	*1*
Calcium	Jang (2000)	Water/Groundwater	mg	2860
Chloride	Jang (2000)	Water/Groundwater	mg	74.0
COD, Chemical Oxygen Demand	Jang (2000)	Water/Groundwater	mg	49.3
Magnesium	Jang (2000)	Water/Groundwater	mg	1.44
Potassium	Jang (2000)	Water/Groundwater	mg	240
Sodium	Jang (2000)	Water/Groundwater	mg	52.0

Table 8-6. Proposed LCI Dataset: Concrete debris, Use as Soil Fill Substitute

Input Flow	Source	Category	Unit	Amount
Demolished concrete, at demolition		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Recycled concrete aggregate, use as soil fill substitute		*Construction and Demolition Debris Management*	*kg*	*1*
Calcium	Jang (2000)	Water/Groundwater	mg	2860
Chloride	Jang (2000)	Water/Groundwater	mg	74.0
COD, Chemical Oxygen Demand	Jang (2000)	Water/Groundwater	mg	49.3
Magnesium	Jang (2000)	Water/Groundwater	mg	1.44
Potassium	Jang (2000)	Water/Groundwater	mg	240
Sodium	Jang (2000)	Water/Groundwater	mg	52.0

8.7 Data Gap Analysis and Opportunities for Additional LCI Data

Table 8-7 summarizes the type of US-based LCI identified from reviewed concrete EOL management sources, including Wilburn and Goonan (1998), Jang (2000), and Cochran (2006). Wilburn and Goonan (1998) and Cochran (2006) only provide partial data only present energy-consumption information for concrete processing. Data sources that are not the primary source of LCI data have not been included in Table 8-7 [e.g., the WARM model is not included because it uses data from Wilburn and Goonan (1998) and is not the original source of data for the energy consumed in processing recycled concrete]. Overall, limited EOL-specific LCI are available for concrete. Based on a review of the available information, the following data gaps were identified for compiling a more comprehensive LCI dataset for concrete EOL management:

1. **Leachable emissions from the landfill disposal or use of RCA as aggregate or in the replacement of natural soils are lacking.** Although an estimate of liquid emissions from concrete disposal in unlined CDD landfills and for use as aggregate and natural soil is included in the proposed LCI datasets and was developed based on batch and column concrete leaching test data reported by one study (Jang 2000), these liquids emissions are associated with a maximum L:S of 20 and do not represent the complete liquid emission perspective. Future research should consider estimating the long-term leaching of RCA (at an L:S ratio that would represent complete contaminants leaching).

2. **The long-term impact of using recycled concrete in pavement applications and effect on quality and service life of the pavement should be assessed**. Numerous studies of the properties of RCA and how it affects concrete and pavements constructed with it are available. Evaluating long-term US-studies comparing RCA pavement durability and service life over an extended period of time would be a valuable contribution to better understanding environmental burdens over the entire life cycle of recycled concrete in pavement applications.

3. **Limited data available for quantifying concrete carbonation**. Studies have shown that the cement portion of concrete can over time absorb carbon dioxide in a process called *carbonation*. There have been several recent studies (e.g., Pade and Guimaraes 2007, Dodoo et al. 2009, Collins 2010, and García-Segura et al. 2014) describing and comparing observations of carbon dioxide uptake in different stages of the life cycle of concrete (e.g., calcination of cement, concrete usage phase, demolition of concrete, use of crushed concrete as an aggregate). Most of these studies estimate carbon dioxide uptake in concrete by using predictive modeling based on Fick's law of diffusion and a carbon uptake equation developed by Lagerblad (2005). This modeling approach incorporates the following parameters into the estimation of carbon dioxide update: the amount of carbonation that has already occurred in the concrete, the amount of Portland cement in the concrete, the amount of calcium oxide in cement, the molar weight of oxide (carbon dioxide/calcium oxide), the service life of the concrete, the exposed concrete surface area, and a carbonation rate coefficient based on the strength and environmental exposure conditions of the concrete. The EOL management of concrete has been identified as an important component in calculating carbon dioxide emissions in the life cycle of concrete (Collins 2010). Concrete EOL management may enhance carbonation, particularly in recycling applications because concrete is commonly size reduced when it is recycled; this creates a greater surface area that exposes fresh uncarbonated carbon for carbonation. Due to lack of data, a carbonation factor was not included in the proposed LCI.

4. **Differences in transportation emissions between mobile concrete processing operations and stationary concrete processing facilities**. Recycling of concrete can occur at the site of demolition (e.g., a concrete pavement being demolished and then crushed for use as subbase onsite) and a

mobile processing unit, usually smaller than a concrete processing facility, can be brought to the construction site. The differences in using materials at the site of demolition will likely be realized in emissions savings in transportation when comparing the emissions from the transportation necessary to mobilize the processing equipment to the site and back and loading up and trucking large amounts of material to a stationary facility. Details on the distances mobile operations are transported, how they are transported, and average distances and the methods of transporting (e.g., barge, trucks, train) concrete to a recycling facility would be valuable in improving transportation emissions estimates to compare each processing option.

5. **LCI for mobile and stationary concrete processing facilities**. Mobile processing units are usually smaller and likely less efficient than a stationary concrete processing facility. Two sources were identified that estimated the amount of energy necessary to process concrete [Wilburn and Goonan (1998) and Cochran (2006)]. Cochran's (2006) energy consumption estimate was based on a survey of equipment manufacturers and only includes fuel consumption in equipment. Wilburn and Goonan (1998) did not provide sufficient detail for activities and operations included in the reported energy requirement estimate. Neither of these sources provided a breakdown of fuel types (e.g. electricity, diesel). An estimate of the average fuel mix used for concrete processing is necessary for developing a more accurate LCI dataset for concrete processing. Also, data for particulate matter emissions released from concrete processing as well as water usage for controlling particulate matter emissions are lacking. No data pertaining to the environmental burdens associated with the manufacture or decommissioning of concrete processing equipment were identified.

6. **No data were found to assess the difference in the energy requirement for processing reinforced and non-reinforced concrete**. Amounts of waste produced from processing concrete were not identified. Although a majority of non-reinforced concrete may result in very little waste to be disposed of, the recovery rate of concrete from reinforced concrete is likely to be less because concrete can remain stuck to steel mesh or rebar. Amounts of recovered rebar per unit mass of processed reinforced concrete were also not identified.

Table 8-7. Overview of LCI Data Available for Concrete

Process	Wilburn and Goonan (1998)	Jang (2000)	Cochran (2006)
Concrete Removal/Demolition			
Transport	P		
Landfill Leachate		X	
Demolished Concrete Processing	P		P
RCA Replacement of Primary Aggregate			
Concrete Debris Replacement of Natural Soil			

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9 Recovered Screened Material

9.1 Introduction

RSM, sometimes referred to as CDD fines, is a by-product of CDD material recovery (i.e. processing) operations. RSM includes soil, sand, and small aggregates from land clearing and demolition, as well as small particles of larger CDD materials that break off during material handling and sorting (e.g., gypsum drywall). The actual gradation of the RSM material will depend on the screen size(s) used during CDD processing, where common screen sizes include openings from 0.6 to 5 cm (Jang and Townsend 2001a). RSM EOL management options may include landfill disposal or recycling options, including use as landfill alternative daily cover (ADC) or application as a general fill. Both recycling management options would replace the primary production and use of natural soil resources. Figure 9-1 shows a flow diagram depicting EOL management processes for RSM.

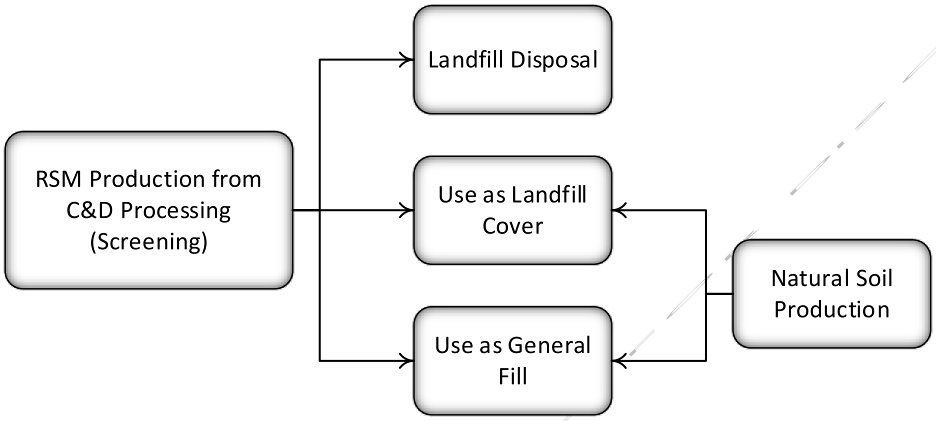

Figure 9-1. Material Flows for RSM Production and EOL Management

Figure 9-2 presents the composition of RSM reported by Townsend et al. (1998), who characterized RSM samples collected from 13 CDD processing facilities in Florida. While the majority of RSM resembled a soil-like material (less than 0.64 cm in size), RSM fraction retained on 0.64-cm screen (referred to herein as an *identifiable* fraction) was further segregated into individual components. The identifiable fraction represented approximately 27% of RSM. As is evidenced in the figure below, the major CDD material categories in identifiable RSM include small pieces of aggregate (e.g., rock, concrete), wood, ceramics, paper, and drywall; these materials combined account for over 75% of the identifiable fraction. Of particular interest is the drywall component of RSM. As discussed in the drywall chapter, the placement of drywall in anaerobic conditions contributes to the production and release of hydrogen sulfide gas.

Currently, there is no known estimate of the nationwide production of RSM. A summary of recycled CDD materials as provided by four states (Massachusetts, Florida, Washington, and Nevada) in 2011/2012 suggests that the fines content of recycled CDD materials is approximately 8% (US EPA 2014). However, other sources suggest that fines represent nearly 25% of the mass of material handled at CDD processing facilities (Calhoun 2012, Jang and Townsend 2001a, 2001b). Therefore, total nationwide RSM production is estimated to range from 3.7 to 11.9 MMT of material.

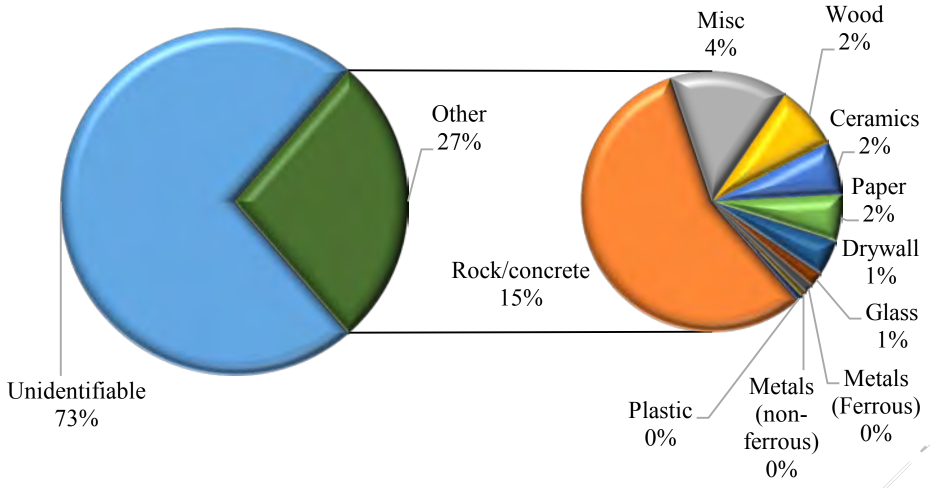

Figure 9-2. Composition of RSM (Townsend et al. 1998)

9.2 EOL Management

RSM is produced as a result of the EOL management of other CDD materials. As it is a byproduct of CDD processing, RSM carries no upstream emission burdens. Once recovered from CDD materials processing, RSM is either disposed of at a landfill, used as ADC at a landfill, or potentially applied in a general fill application. There is currently no known nationwide estimate of the quantity of material that is handled through each management option. Table 9-1 lists the processes that should be considered in developing an LCA for the EOL management of RSM.

Table 9-1. RSM EOL Management Process Descriptions

Process	Description
Landfill Disposal	Depending on state and local regulations, RSM may be disposed of in either a CDD or a MSW landfill.
RSM use as ADC	RSM may be used in place of soil cover material at a landfill site. The use of RSM in this application would preclude the emissions resulting from the excavation and transport of natural soil on/to the landfill site.
RSM use as General Fill	RSM may be used in place of soil in a general fill application. The use of RSM in this application would avoid the emissions resulting from the production and transport of natural soil to the general fill site.
Transport	While transport LCI are generally presented in units of mass-distance (kg-km), regional average distances between processes is necessary for developing a regional-level LCA. It should be noted that cover soil production may occur at the landfill site where the soil will be applied.

9.3 LCI Sources

Government publications, peer-reviewed literature, and LCA models were searched for LCI datasets pertaining to RSM management. None of the LCA models reviewed contained emissions inventory data on RSM (or CDD fines). While Doka (2009) discusses the EOL management of fines resulting from the operation of building waste sorting facilities, no information/datasets were found in the report or from

Ecoinvent database search queries that provide emissions data. Table 9-2 lists the sources identified to contain information or data which could be used for development LCI for RSM management processes within an LCA framework.

Table 9-2. List of Sources Reviewed for LCI Data

LCI Source	Description
Townsend et al. (1998)	The total and (SPLP) leachable concentrations of heavy metals, VOCs, and semi-VOCs are reported from RSM samples collected from 13 CDD processing facilities over 1996-1997. Additional sources that use the same raw data to perform supplementary studies include Jang and Townsend (2001a, 2001b) (organic compound leaching and sulfate leaching, respectively) and Townsend et al. (2004) (heavy metals).
Anderson et al. (2010)	Hydrogen sulfide data from six MSW landfill sites were used to estimate site-specific gas generation potential (volume of hydrogen sulfide produced per ton sulfur) and first-order decay rates. Eighty-nine samples of CDD fines were collected from CDD processing facilities and landfill sites to estimate the total sulfur content of CDD fines.
Tolaymat et al. (2013)	Estimated laboratory hydrogen sulfide decay-rate constants for paperless drywall, three sizes of crushed drywall (with paper), and estimated a theoretical hydrogen sulfide generation potential per mass of drywall.
US EPA (2012)	WARM has compiled estimates of LFG emissions (i.e. methane and carbon dioxide) for numerous CDD waste materials (e.g., paper, wood, drywall) using numerous sources. These estimates are used in conjunction with identified material fractions in RSM to develop a weighted estimate of total RSM LFG emissions.
Ecoinvent	Doka (2009) (background documentation for the Ecoinvent LCI database) discusses the management of various building materials, including CDD fines.

9.4 LCI Related to Production

As mentioned previously, RSM is a byproduct (i.e. residual) of CDD processing and is a composite of various CDD materials that readily fragment into smaller pieces during the CDD material handling at materials recovery facility. As RSM recovery is not the primary objective of the CDD materials processing, the emissions associated with CDD processing should not be allocated to the RSM. In other words, emissions associated with the construction, operation, and decommissioning of the CDD processing facility should be allocated to the individual CDD materials targeted for recovery.

9.5 LCI Related to Disposal

RSM may be disposed of in either a CDD or an MSW landfill, where both leachate and gas emissions will result. While it would be possible to take individual leaching data for each of the individual materials and use the mass-fraction information as presented in Figure 9-2 to develop an estimate of the composite leachability of RSM, this approach could potentially introduce emission inaccuracies as a result of the following:

- No estimate of the composition of the unidentifiable fraction of RSM is available. Approximately a quarter of RSM was identified as specific CDD materials. While the identified fractions of RSM

could be applied to the unidentified and "miscellaneous" portion of the material on an equal mass-fraction basis, the majority of the identified material likely consists of soil and small aggregate material which may have substantially different leaching properties.
- There may be synergistic or antagonistic leaching effects between individual components for RSM. For example, alkaline materials (e.g. gypsum drywall) may have an immobilizing effect on the leaching of some heavy metals.
- There is very limited information on the long-term leachability of specific CDD materials.

Ecoinvent considers disposal of fines from CDD recovery facilities in landfills (Doka 2009). Placement in a sanitary landfill is the typical practice due to challenges for recyclers resulting from the presence of chlorides, sulfates, and fibers in the material. The fines can be disposed of in an inert or residual material landfill depending on its properties, including the amount of materials that can dissolve when mixed with distilled water. However, the solubility of the gypsum portion of the fine material alone is typically sufficient to require placement in a sanitary landfill.

9.5.1 Leachable Emissions from RSM

Townsend et al. (1998) conducted a characterization of chemical and engineering properties (e.g., grain size distribution, friction angle) of RSM using 99 samples collected from 13 CDD processing facilities in Florida from 1996 to 1997. The total and leachable concentration of VOCs, semi-VOCs, and heavy metals in RSM were analyzed; leaching data were based on SPLP tests. Townsend et al. (2004) provides the mean concentration for each metal (not including BDL readings) and the number of detected and non-detected measurements. The mean metal concentration measurement was adjusted for non-detects by including the respective detection limit as the concentration for measurements below detection limit readings. An average measured concentration was only developed for the metals that were detected in more than half of all the samples analyzed, i.e., aluminum, arsenic, and zinc.

Average leachable amounts of VOC and semi-VOC concentrations were estimated using the raw data presented by Townsend et al. (1998). Average parameter measurements were only developed for those parameters which were detected in over half of all the samples analyzed. For these parameters, non-detect measurements were included as the minimum quantification limit. Results from the SPLP tests were converted from volume-based concentrations (i.e., leached parameter mass per solution volume) to mass-based concentrations to (i.e., leached parameter mass per RSM mass) concentrations by multiply the volume-based concentration by the SPLP L:S and adjusting for the moisture content of the sample.

Because leachable emissions from RSM were estimated using the SPLP testing procedure, the estimates only simulate the leaching for applications such as disposal in inert debris landfills or general fill where contaminants leach due to exposure to precipitation. TCLP simulates the contaminants leaching in the biogeochemical environment of an MSW landfill.

9.5.2 Landfill Gas Emissions for RSM

The organic fraction of RSM (e.g., wood, paper) will contribute to the release of carbon dioxide, methane, and other VOCs in an anaerobic environment. US EPA (2012) summarizes information provided by Barlaz et al. (1989), Eleazer et al. (1997), and Barlaz (1998) to estimate methane and carbon dioxide emission factors for the landfill disposal of individual RSM materials, including wood, paper and drywall. Additional data on the development of these emission factors for MSW and CDD materials landfill disposal is included in Chapter 2. The individual emission factors for each material were multiplied by the RSM material mass fractions (presented in Figure 9-2) and summed to develop a composite emissions factor for each gas due to the absence of emission factors for methane and carbon dioxide for RSM in literature. It was assumed that the "unidentifiable fraction" and the "miscellaneous" portion of the "identifiable" fraction of RSM will not contribute to LFG production, as a significant portion of these fractions likely represent sand, soil, small aggregate, and other non- or poorly-degradable materials. While RSM is produced from CDD processing facilities, landfill placement of RSM may occur at either CDD landfills or at MSW landfills. Because of cardboard's prevalence in building material packaging, it is assumed that gas emissions data for cardboard is representative of that from the degradation of the paper fraction encountered in RSM (as presented in Figure 9-2).

Table 9-3 presents carbon dioxide and methane emission factors associated with disposal of wood (i.e. dimensional lumber), cardboard and drywall and provides the mass fraction of these materials in RSM to estimate the resulting methane and carbon dioxide emissions from RSM disposal in CDD and MSW landfills. As discussed in Chapter 2, Section 2.5.10.8, these emission factors include 10% oxidation of uncollected methane into carbon dioxide.

Table 9-3. Methane and Carbon Dioxide Emission Factors for CDD and MSW Landfill Disposal of Different RSM Components (developed from US EPA (2012))

Material	Fraction of RSM (w/w) (Jang and Townsend 2001b)	Emissions from CDD Landfills		Emissions from MSW Landfills	
		Methane Emissions (kg/kg wet material)	Carbon Dioxide emissions (kg/kg wet material)	Methane Emissions (kg/kg wet material)	Carbon Dioxide emissions (kg/kg wet material)
Wood	2.19%	0.064	0.21	0.022	0.33
Cardboard	1.59%	0.12	0.40	0.042	0.60
Drywall	1.21%	0.010	0.034	0.0038	0.052
RSM	**-**	**0.003**	**0.011**	**0.0012**	**0.017**

In addition to the release of methane and carbon dioxide (from the decomposition of drywall paper facings), the drywall component of RSM will contribute to the production of hydrogen sulfide gas if placed in an anaerobic environment (Jang and Townsend 2001b, Lee et al. 2006). Anderson et al. (2010) used LFG hydrogen sulfide concentration data, waste tonnage records, and LFG flow rates from six MSW landfill sites to estimate the site-specific hydrogen sulfide gas generation potential (cubic feet hydrogen sulfide /ton sulfur) and first-order decay rates of RSM drywall. Anderson et al. (2010) also collected 89 samples of CDD fines from CDD processing facilities and landfill sites to estimate a total average CDD fines sulfate content of 4.3% or a sulfur content of 1.4% (by weight). Therefore, approximately 14 grams of sulfur would be disposed with every kilogram of RSM. The average hydrogen sulfide generation potential from four landfill sites that did not monofill CDD fines (i.e., separately place CDD fines in a different location than incoming MSW) was presented as 5,360 ft^3 of hydrogen sulfide per ton of landfilled sulfur. Using a density of 1.42 grams per liter of hydrogen sulfide at 20^o C and 1 atmosphere pressure, this approximately equates to 3.4 grams of hydrogen sulfide generation per kilogram of RSM disposed. The actual generation of hydrogen sulfide at a specific site depends on a host of factors (e.g. precipitation, waste composition, presence of organic matter, cover soil application, and compaction practices). The hydrogen sulfide generation potential across the four sites described above ranged from 3,186 to 7,634 ft^3 of hydrogen sulfide per ton of landfilled sulfur.

Plaza et al. (2007) performed a laboratory-scale study using drywall degradation in columns to evaluate the hydrogen sulfide attenuation efficiency of a 15-cm-thick layer of five landfill cover materials, including fine crushed concrete, coarse crushed concrete, soil amended with lime, sandy soil, and clayey soil. The sandy and clayey soils were natural soils taken in the vicinity of the lab and were respectively found to have hydrogen sulfide removal efficiencies of 29.7% and 65.3% compared to control columns where no cover material was used. An average of these removal efficiencies (47.5%) was used to simulate hydrogen sulfide attenuation for RSM disposal at an average landfill site. As is evident by the range of hydrogen sulfide removal efficiencies from naturally-occurring cover soils, actual removal efficiencies are strongly dependent on the type of soil used. It should also be noted that a 15-cm-thick layer of cover soil is more representative of the type of cover soil which would be used daily at an MSW landfill and often weekly (or more or less frequently) at a CDD landfill. Final and intermediate landfill covers will be thicker and would likely increase hydrogen sulfide attenuation. However, a majority of the hydrogen sulfide emissions would have occurred prior to final cover placement due to high hydrogen sulfide generation rates. Anderson et al. (2010) estimated a relatively high hydrogen sulfide generation rate constant, ranging from approximately 0.5 to 0.9 year^{-1}. Using the average decay rate constant of 0.702 year^{-1} (calculated from landfills that did not monofill CDD fines) approximately 75% and 97% of the total hydrogen sulfide emission would occur within 2 and 5 years, respectively. The final cover, therefore, would only have a limited role in attenuating hydrogen sulfide emission and the vast majority of hydrogen sulfide gas would likely be mitigated through daily/weekly covers only.

The GCCS is also estimated to have a limited role in hydrogen sulfide emission attenuation. According to federal regulations, the gas collection deadline for MSW landfill disposal areas is 2 years for areas that have reached final grade and 5 years for those areas that have not reached final grade. Therefore, LFG collection is expected to have a negligible impact on hydrogen sulfide emission from the disposal of RSM at MSW landfills. The same emissions factor for hydrogen sulfide is assumed for both CDD and MSW landfill sites in the absence of additional data. Therefore, including the effects of cover soil attenuation, it is estimated that 0.0018 kg of hydrogen sulfide will be released for every kg of RSM disposed of in a landfill.

Table 9-4 and Table 9-5 present the emission and material burdens associated with CDD and MSW landfill disposal of RSM, respectively. As described previously, leachable emissions from RSM disposed of in an MSW landfill are not provided due to the lack of TCLP leaching data. Details describing the calculations for estimating the quantity of cover soil used and diesel and electricity consumed for operating CDD and MSW landfills, and the materials and energy requirements for the construction and closure and post-closure care of MSW landfills are provided in Chapter 2 of this report. The bulk density of RSM as provided by Jang and Townsend (2001b) was used to estimate landfill cover soil requirements. In the absence of an average nationwide transport distance between CDD processing facilities and CDD and MSW landfills, a distance of 20 km was assumed.

Table 9-4. Proposed LCI Dataset: Recovered Screened Material, at Unlined CDD Landfill

Input Flow	Source	Category	Unit	Amount
Recovered screen material, from CDD processing		Construction and Demolition Debris Management	kg	1
Cover soil, from offsite source	See Chapter 2	Construction and Demolition Debris Management	kg	0.0140
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
CDD landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Aluminum	Townsend et al. (2004)	water/groundwater	mg	929

Arsenic	Townsend et al. (2004)	water/groundwater	mg	176
Benzene, ethyl	Townsend et al. (1998)	water/groundwater	µg	56
Calcium	Jang and Townsend (2001b)	water/groundwater	g	12
Carbon Dioxide	US EPA (2012)	air/unspecified	kg	0.011
Hydrogen Sulfide	Anderson et al. (2010)	air/unspecified	kg	0.0018
Methane	US EPA (2012)	air/unspecified	kg	0.0034
Phthalate, bis(2-Ethylhexyl)	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	mg	0.072
Phthalate, di-n-Butyl	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	mg	0.062
Recovered screened material, at unlined CDD landfill		*Construction and Demolition Debris Management*	*kg*	*1*
Sulfate	Jang and Townsend (2001b)	water/groundwater	g	30
Toluene	Townsend et al. (1998)	water/groundwater	µg	403
Toluene, 4-Isopropyl	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	34
Trichlorofluromethane	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	289
Trimethylbenzene, 1,2,4-	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	38
Xylene, m/p-	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	256
Xylene, o-	Townsend et al. (1998)	water/groundwater	µg	122
Zinc	Townsend et al. (2004)	water/groundwater	mg	1330

Table 9-5. Proposed LCI Dataset: Recovered Screened Material, at MSW Landfill

Input Flow	**Source**	**Category**	**Unit**	**Amount**
Recovered screen material, from CDD processing		Construction and Demolition Debris Management	kg	1
Cover soil, from MSW landfill stockpile	See Chapter 2	Construction and Demolition Debris Management	kg	0.098
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75	Assumed		t*km	0.001*20
MSW landfill construction, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill operations	See Chapter 2	Construction and Demolition Debris Management	kg	1
MSW landfill closure and post-closure, for CDD materials	See Chapter 2	Construction and Demolition Debris Management	kg	1
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Carbon Dioxide	US EPA (2012)	air/unspecified	kg	0.017

Hydrogen Sulfide	Anderson et al. (2010)	air/unspecified	kg	0.0018
Methane	US EPA (2012)	air/unspecified	kg	0.0012
Recovered screened material, at MSW landfill		*Construction and Demolition Debris Management*	*kg*	*1*

9.6 LCI Related to Recycling

The primary beneficial uses of RSM include use as an ADC and use in a general fill application. Both beneficial uses result in avoidance of the production of natural soil. Soil for use at general fill sites is typically excavated, transported, and immediately used by the end user. The soil for landfill cover use is often excavated and temporarily stockpiled onsite before use. The details of LCI proposed for natural soil excavation, on-site transport, and stockpile (for landfill cover) are presented in Chapter 2.

While RSM has properties that could make it favorable for use in place of traditional soil material (e.g., potentially improved drainage and traction during rain), use as an ADC in an anaerobic environment presents the same hydrogen sulfide production challenges as landfill disposal (Carlton et al. 2005, Musson et al. 2008). Except for cover soil and landfill operation requirements (i.e., electricity and diesel consumption for site operation is completely allocated to disposed materials, not cover materials), the use of RSM as ADC is expected to have the same leachate and gas emissions as RSM placed in a landfill for disposal. Both disposal sites are assumed to be located 20 kilometers from RSM production.

RSM may be beneficially used in place of natural soil for a general fill. Clark et al. (2010) presents a case study examining the use of RSM in Florida for grading 60 residential sites located in low-lying areas. RSM was mixed with onsite soil and placed at these residences to help alleviate historic flooding problems. Following placement of the material, several of the property owners expressed concern regarding the potential leaching of RSM contaminants. A follow-up investigation suggested that while the arsenic and Total Recoverable Petroleum Hydrocarbon (TRPH) concentrations were elevated above Florida soil cleanup target levels, the concentrations were either in the range of or below area background soil concentrations and were not at levels which presented a public health threat.

The proposed LCI dataset for the use of RSM as a general fill is presented as Table 9-6. This table includes the same set of leachable emissions as was estimated for disposal of RSM in a CDD landfill. However, it is assumed that anaerobic conditions would not develop in the RSM-based general fill, so gas emissions are not included. The burdens associated with RSM placement and surface grading for an "average" fill application are unknown and not included in the dataset. The general fill site is assumed to be located 20 km from RSM production.

Table 9-6. Proposed LCI Dataset: Recovered Screened Material, Use in Environment

Input Flow	Source	Category	Unit	Amount
Recovered screen material, from CDD processing		Construction and Demolition Debris Management	kg	1
Truck transport, class 8, heavy heavy-duty (HHD), diesel, short-haul, load factor 0.75			t*km	0.001*20
Output Flow	**Source**	**Category**	**Unit**	**Amount**
Aluminum	Townsend et al. (2004)	water/groundwater	mg	929
Arsenic	Townsend et al. (2004)	water/groundwater	mg	176
Benzene, ethyl	Townsend et al. (1998)	water/groundwater	µg	56
Calcium	Jang and Townsend (2001b)	water/groundwater	g	12
o-Xylene	Townsend et al. (1998)	water/groundwater	µg	122
Phthalate, bis(2-Ethylhexyl)	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	mg	0.072
Phthalate, di-n-Butyl	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	mg	0.062
Recovered screen material, use in environment		*Construction and Demolition Debris Management*	*kg*	*1*
Sulfate	Jang and Townsend (2001b)	water/groundwater	g	30
Toluene	Townsend et al. (1998)	water/groundwater	µg	403
Toluene, 4-Isopropyl	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	34
Trichlorofluromethane	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	289
Trimethylbenzene, 1,2,4-	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	38
Xylene, m/p-	Townsend et al. (1998)	Construction and Demolition Debris Management/groundwater	µg	256
Zinc	Townsend et al. (2004)	water/groundwater	mg	1,330

9.7 Data Gap Analysis and Opportunities for Additional LCI Data

Table 9-7 summarizes the extent of the LCI information found for developing the processes discussed in this chapter. All the data sources found provided information specific to the US. As shown in Table 9-7, all LCI information found for the management of RSM are partial. WARM LFG emissions are partial because US EPA (2012) only focuses on major greenhouse gas (GHG) emissions from different components of RSM – emission estimates for non- and minor GHGs and other LFG emissions are not included (e.g. hydrogen sulfide, VOCs) . Townsend et al. (1998) and related sources only provide partial data because non-metal inorganics were not included in the analysis, and only batch SPLP data are available. Anderson et al. (2010) and Tolaymat et al. (2013) specifically focus on hydrogen sulfide generation and do not provide emission estimates for other gaseous constituents related to RSM. Based on a review of currently available LCI data on RSM management, the following data gaps were identified for RSM process datasets:

1. **Long-term leachable emissions from RSM placed in a CDD or an MSW landfill.** While Townsend et al. (1998) and related investigations (see Table 9-2) conducted an extensive characterization of RSM from numerous CDD processing facilities from across the state, including leaching of heavy metals, VOCs, and semi-VOCs emissions, sample leachability was assessed over for a L:S ratio of 20. The leaching emission estimates represent only partial leaching amounts. Jang and Townsend (2001b) provide very limited information on SPLP concentrations of non-metal inorganics (e.g. sulfate, calcium). Moreover, SPLP data simulate contaminant leaching in inert debris landfills (e.g., CDD landfill) or land application scenarios and cannot be extended to estimate leaching in biogeochemical environment of an MSW landfill.

2. **Gaseous emissions from the anaerobic decay of RSM.** No gaseous emission measurements associated with decomposition of RSM in anaerobic environments are reported in the literature. Methane and carbon dioxide emissions estimates presented above are based on the mass fraction of specific organic constituents in RSM. Reported hydrogen sulfide measurements have either been measured from the decay of composite MSW (which may have other materials contributing to the release of hydrogen sulfide) or from the individual decay of drywall. Gaseous emission measurements from the actual material would provide a better estimate than measurements of individual components of the material.

3. **Average nationwide transport distances between CDD processing facilities and disposal/beneficial use sites.** These distances are of particular importance for the development of comparative LCA to analyze the potential benefit or burden of beneficially using RSM as a substitute for natural soil.

Table 9-7. Overview of LCI Data Available

Process	WARM	Townsend et al. (1998) and related	Anderson et al. (2010)	Tolaymat et al. (2013)
Landfill Gas Emissions	P		P	P
Landfill Leachate Emissions		P		
General Fill Leachate Emissions		P		

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10 Clay Bricks

10.1 Introduction

Clay bricks represent a relatively small fraction of the total CDD material stream and are commonly generated from the demolition of buildings, structures, and pavements. The quantity of clay bricks produced from demolition will vary depending on the building composition. Most clay bricks are produced from common clay and shale, a material obtained from mining and excavation which must go through an extensive drying and extrusion process prior to kiln firing. Clay bricks are primarily manufactured for structural construction purposes, which require the use of common face brick, with over 3.3 billion clay bricks being produced in 2008 in the US; this quantity accounts for 60% of the nationwide total annual production of all brick types (USCB 2011). According to USGS, close to 15.9 MMT of common clay was mined and nearly 57% of this was used for clay brick production (USGS 2008). Figure 10-1 shows the typical flow of clay bricks from the primary extraction and processing through the EOL management of clay bricks. The processes from the EOL removal of clay brick to their ultimate beneficial reuse or landfill disposal should all be considered for conducting an LCA for EOL management.

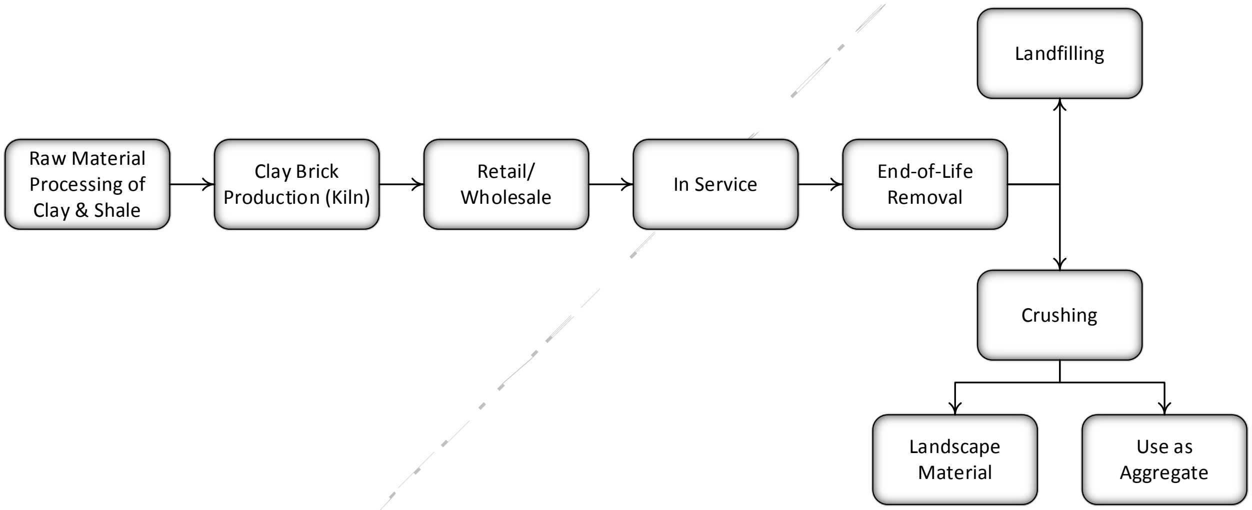

Figure 10-1. Material Flows for Production through EOL Management of Clay Bricks

The clay extraction and brick manufacturing processes are not presented in detail in this chapter as clay bricks are not recycled in closed loop (US EPA 2012a). The emissions associated with processing clay bricks to produce recycled aggregate should be considered for EOL LCA.

10.2 EOL Management

US EPA (2012b) estimates that 1 to 5% of CDD is comprised of bricks. Approximately 136 MMT of CDD were generated in 2011 (US EPA 2014), suggesting that approximately 1.4 to 6.8 MMT of bricks were discarded in 2011. Disposal appears to be the dominant EOL management option for clay bricks. Only limited amounts of discarded clay bricks are recycled. Due to concerns about structural strength, reuse of salvaged bricks in load-bearing applications is not recommended (Webster 2002); salvaged clay bricks are sometime reused in non-structural application such as brick fireplaces, hearths, patios, and other uses (US EPA 2012a). Reza (2013) and Cavelline (2012) reported that typical brick recycling practices include reuse as a replacement for aggregate in structural fills or pavements. Recovered clay bricks should be processed prior to use as aggregate.

Table 10-1 lists the processes that should be considered for conducting an LCA of EOL management options for clay bricks. Primary aggregate production and general material transport LCI datasets are relevant for multiple CDD materials in this report and are presented and discussed in detail in Chapter 2.

Table 10-1. Clay Brick EOL Management Process Descriptions

Process	Description
Building Demolition	The material and energy inputs as well as process non-energy emissions (e.g., particulate matter emission) associated with building demolition should be considered.
Landfill Disposal	Landfilling entails the placement and compaction of bricks and their long-term physiochemical decomposition in a landfill environment.
Clay Brick Processing	Discarded clay brick processing may include sorting, crushing, and fractionation (i.e., sorting into different size categories).
Crushed Clay Brick Use as Aggregate	Recycled aggregate produced from clay bricks may be used as a primary aggregate substitute in a fill application. Primary aggregate production and transport emissions would be avoided with the use of clay bricks as an aggregate material.

10.3 LCI Sources

Peer-reviewed literature and government and private industry publications were reviewed to identify available LCI datasets pertaining to clay brick EOL management. WARM documentation was the only source of US-based data found for EOL management-related data for bricks. Table 10-2 lists data sources reviewed to compile LCI presented in this chapter. If LCI data were not available, process metadata and documentation were reviewed to evaluate the completeness of applicable datasets (e.g., which emissions categories were included, background data used to compile the dataset, geographic location, and time period of the data). The primary sources of information used to develop the LCI datasets and information identified, if available, were reviewed. It should be noted that the sources presenting clay brick production LCI are not listed in the table as clay bricks recycling in closed loop is not prevalent.

Table 10-2. List of Sources Reviewed

LCI Source	Description
Karius and Hamer (2001)	This German study compares total concentration and leaching data from bricks made with 50% dredged harbor sediment to leaching data from manufactured bricks pulled from other European brick companies.
US EPA (2012a)	The WARM Model presents data on GHG emissions associated with the source reduction, transport, and landfilling (i.e., collection and placement) of clay bricks. While clay brick recycling was mentioned, no LCI data were presented.
Ecoinvent	Ecoinvent is an LCI database developed by the Swiss Centre for Life Cycle Inventories, which includes specific processes related to the EOL management of numerous individual materials.

10.4 LCI Related to Disposal

The primary EOL management method for bricks in the US is landfill disposal. While the Ecoinvent (2014) database includes processes for handling waste bricks, the model assumes the placement of bricks in an inert debris landfill for which leachate emissions are not included (Doka 2009). Similar to all the other materials modeled in WARM, US EPA (2012a) estimates GHG emissions released from fossil fuel combustion resulting from the transport to and placement of demolished clay bricks at a landfill. Since clay bricks are an inert material, they do not undergo biological decomposition.

A literature review only yielded international leaching test results for clay bricks. Karius and Hamer (2001) conducted a series of leaching tests (pH static tests with a L:S-ratio of 10) on bricks made from clay and dredging sediments from local water body to assess the impact of the sediments on leaching of 20 contaminants, including metals and sulfur. The tests were conducted on crushed bricks (grain size ranged from 50 to 30,000 µm) made with and without sediment. In general, the heavy metals from the sediment bricks leached at the upper end of the concentration ranges for the bricks made without sediments.

While the transportation, diesel, and electricity requirements for the landfill disposal of clay bricks would be the same on a mass-fraction basis regardless of the type of material, due to lack of data the liquids emissions from brick disposal in a landfill are unknown and an LCI dataset was not developed.

10.5 LCI Related to Recycling

10.5.1 Clay Brick Demolition

Doka (2009) presented building materials demolition-, recycling-, and disposal-related LCI as part of the Ecoinvent database based on management practices in Switzerland. Energy consumption and air emission estimates from the study were derived from other studies that were done in the European Union. Equipment demolition efficiencies (i.e., the time spent per volume of waste demolished) and fuel consumption rates were compiled from literature to estimate the energy required to demolish brick wall, gypsum board, and cement-fiber slab as 0.0359 MJ/kg. Particulate matter is a major non-fuel air emission associated with demolition activities. Doka (2009) included an air emissions factor of 80 mg PM_{10}/kg of demolition waste for all building construction, demolition, and renovations activities. AP-42 provides air emission factor calculation methods for various heavy construction operations, which include dust generation activities from the demolition of buildings and removal of debris as a function of various factors such as site-specific conditions and equipment used (US EPA 1995).

10.5.2 Clay Brick Sorting

Prior to recycling, discarded clay bricks in the mixed CDD waste stream would undergo sorting operations in which bricks are separated from other materials. While Doka (2009) reported energy requirements of CDD materials sorting and size reduction specific to European practices, US-specific data regarding brick processing are lacking. Please see Chapter 2 for more details regarding the development of an LCI dataset for modeling and allocating the environmental burdens of a mixed CDD processing facility.

10.5.3 Clay Brick Use as Aggregate

While clay brick recycling does not appear to be heavily practiced in the US, limited beneficial applications of recovered clay brick as aggregate have been documented (Cavalline 2012, Reza 2013). The Minnesota Department of Transportation conducted a study in 2013 to test the feasibility of reusing bricks in aggregate for road base; the study showed the material met department specifications (Reza 2013). Cavalline (2012) explored the potential use of clay brick rubble as a possible replacement for aggregate in building and pavement concrete mixtures with a focus on the mechanical and engineering properties of brick aggregate. Cavalline (2012) reported that recycled brick may provide acceptable performance when used in pavement and shows promise for use in structural applications. The recovered bricks would need to be size reduced for use as aggregate. US-specific energy requirement and emission data specific to clay bricks processing are lacking.

Similar to other CDD materials that may be processed and beneficially reused as aggregate, the use of demolished clay brick aggregate would offset the production and transport of primary aggregate materials. LCI datasets for primary aggregate production are presented and detailed in Chapter 2 of this report. Due to the lack of gaseous and liquid emission and energy requirement data, LCI for processing and use of clay bricks as recycled aggregates were not developed.

10.6 Data Gap Analysis and Opportunities for Additional LCI Data

Most LCI information on the EOL management of clay bricks is not specific to practices in the US; the US EPA (2012a) was the only source of information that provided US-specific data, but these data only included GHG emissions factors for source reduction and landfill of clay bricks. Based on a review of government publications, peer-reviewed literature, and industry data, the following US-specific LCI data gaps were identified with respect to bricks EOL management:

1. **Energy requirements for sorting/processing clay bricks at a CDD processing facility.** Clay bricks that are part of a mixed CDD stream would be recovered at a CDD processing facility; recovery may include separating, grinding, and fractioning operations (depending on the end-use market). Although diesel consumption data for typical CDD materials sorting operation are available (presented in Chapter 2), size-reduction processing energy requirements and associated emissions specific to brick are lacking.

2. **Long-term leachable emissions from bricks placed as aggregate (e.g., in a fill) or in a landfill.** No US-specific leaching data from clay brick are available to estimate liquids emission from brick placement in a landfill or from beneficial use application as aggregate or fill material.

10.7 References

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11 Summary and Future Research Needs

11.1 Summary

The objective of the work presented in this report was to assess the body of knowledge regarding CDD life-cycle data and to compile US-specific LCI for distinct CDD material categories from publicly available sources. In the previous chapters, available LCI data for processes common to all CDD (Chapter 2) as well as LCI data for eight specific CDD materials (Chapters 3-10) were presented. While the eight materials examined (asphalt pavement, asphalt shingles, gypsum drywall, wood products, LCD, PCC, RSM, and clay bricks) do not represent every component of CDD, they do comprise more than 95% of the total CDD materials generated annually in the US.

As described in the introduction to this report, CDD has not received the degree of attention with respect to environmental emissions or other life-cycle considerations that other waste streams have. Thus, for many LCI categories, US-specific data were not available from publicly available sources. In addition, some of the available data that were used to develop an LCI category were not complete or were approximated by using LCI data from similar materials. Each chapter thus ends with a description of LCI data gaps.

In this final chapter, the data gaps highlighted in the individual chapters are summarized. Table 11-1 presents a summary of processes; associated energy and materials inputs; as well as the gaseous, liquid and solid emissions included in the LCI datasets developed. An "X" denotes that data are included in the developed LCI category, though it does not indicate that all flows were included and/or quantified, only information that was found for that category. Product manufacturing process LCI were evaluated only for those CDD materials that are currently recycled in a closed loop. Construction materials manufacturing and production LCI that were found to be appropriate for CDD LCI but that are contained in existing US NREL or US EPA LCI datasets were not reproduced in this report.

Table 11-1. Summary of CDD Material LCI Process Datasets and Flows Included in the Report

Material	Process	Energy Input	Material Input	Emissions		
				Air	Water	Land
Granite: Crushed and Broken	Production and Transport	X		X		
Construction Sand and Gravel	Production and Transport	X		X		
Limestone: Crushed and Broken	Production and Transport	X		X		
Other Stone: Crushed and Broken	Production and Transport	X		X		
Natural Soil from Borrow Pit	Excavation	X				
Asphalt Pavement	HMA production	X		X		
	Reclaimed asphalt pavement processing	X				

Material	**Process**	**Energy Input**	**Material Input**	**Emissions**		
				Air	**Water**	**Land**
	Use for HMA production[2]					
	Use as aggregate				X	
	Disposal in CDD landfill	X	X		X	
	Disposal in MSW landfill					
Asphalt Shingles	Use for HMA production[1]					
	Reclaimed asphalt shingle processing	X				
	Use as aggregate				X	
	Use as general fill				X	
	Disposal in CDD landfill	X			X	
	Disposal in MSW landfill					
Gypsum Drywall	Facing and backing paper production	X	X	X	X	X
	Virgin gypsum production	X	X	X	X	X
	1/2" Regular and 5/8" Type X drywall manufacturing	X	X	X	X	X
	Drywall processing-size reduction and screening					
	Use in agricultural application	X			X	
	Disposal in CDD landfill	X	X	X	X	
	Disposal in an MSW landfill					
Wood Products	Mulch production and land application	X			X	
	Combustion with energy recovery[3]					X
	Disposal in CDD landfill (untreated/treated)	X	X	X	X	
	Disposal in MSW landfill (untreated/treated)	X	X	X	X	
	Ash disposal in CDD landfill	X	X		X	
	Ash disposal in MSW landfill (untreated/treated)	X	X		X	
	Ash land application					
LCD	Size reduction	X		X		
	Mulch production and land	X		X	X	

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[2] These processes only represent the use of recycled materials to substitute primary materials – specific emissions for these processes were not quantified.

[3] An LCI dataset for this process is already provided in the US EPA LCI database – however, proposed modifications were made for this process to account for moisture content and ash production

Material	**Process**	**Energy Input**	**Material Input**	**Emissions**		
				Air	**Water**	**Land**
	application					
	Compost production and land application	X		X		
	Disposal in CDD landfill	X	X	X	X	
	Disposal in MSW landfill					
	Open burning	X		X		
	Air curtain incineration	X		X		
	Combustion with energy recovery[2]					X
	Ash disposal CDD landfill	X	X			
	Ash disposal in MSW landfill	X	X			
	Ash land application					
PCC	Use as aggregate				X	
	Size reduction	X				
	Use as general fill				X	
	Disposal in CDD landfill	X	X		X	
	Disposal in MSW landfill					
RSM	Use as general fill				X	
	Disposal/Use as cover in CDD landfill	X	X	X	X	
	Disposal/Use as cover in MSW landfill	X	X	X	X	
Clay Bricks	Use as aggregate					
	Use as general fill					
	Disposal in CDD landfill					
	Disposal in MSW landfill					
Mixed CDD	CDD recovery at processing facility	X[4]				X

11.2 Data Gaps and Future Research Opportunity

Based on the data gaps highlighted earlier in the report and that can be inferred from the table above, future data-gathering and research opportunities have been identified. The following sections highlight major LCI data categories pertaining to CDD materials and summarize their associated data gaps and identified research needs.

EOL management practices of CDD materials

Of all the CDD materials reviewed during this study, only the EOL management practices of asphalt pavement were found to be substantially well-documented and quantified; NAPA has been conducting a

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[4] Only diesel consumption information was found – process electricity use still needs to be assessed.

U.S.-wide annual survey of paving mix producers since 2009 to track uses of asphalt pavement reclaimed from road construction and maintenance projects. Although the USGS also compiles and reports the amounts of RAP- and PCC-derived recycled aggregates in the US based on a survey of aggregate producers and C&D contractors, the data are incomplete because of survey limitations. For other materials, data on the amount of materials managed via different EOL management options (e.g., recycled versus disposed) are very limited.

Many state environmental agencies track the statewide amount of CDD materials landfilled annually, but only four states (Florida, Massachusetts, Nevada, and Washington) appear to closely track the amount of materials recycled. Due to this lack of EOL management recycling data and because of the interest of multiple government agencies (e.g., USGS, U.S. DOE, U.S. EPA, FHWA, state environmental and highway) and industry organizations (e.g., NAPA, CDRA) in analyzing this information, there is an opportunity for collaborative research on the quantification of CDD materials EOL management practices.

CDD Material Processing LCI

CDD materials require some degree of processing prior to use in open- or closed-loop recycling applications. CDD materials processing facilities require energy (e.g., electricity, diesel) and material (steel used for building and equipment used for processing) inputs and release process energy and non-energy emissions. Only a few of sources were identified to have reported CDD materials processing energy requirements (Wilburn and Goonan 1998; Cochran 2006). Moreover, the available estimates are based on limited data.

It is expected that most CDD material processors track basic energy demands (e.g., daily, weekly, or monthly fuel and electricity usage) and material input data. If compiled, these data could provide valuable input for more reliable estimates of CDD material processing energy requirements. The CDRA and the University of Florida are currently conducting a nationwide survey of CDD materials recyclers to characterize CDD recycling facilities in terms of material throughput, jobs, and energy use (among others). Several federal agencies (e.g., the U.S. EIA and the U.S. Census Bureau) routinely survey U.S. industries to collect various economic and labor data – addition of material and energy usage could potentially supplement these existing surveys, and the results could be used to develop LCI. Although a NAICS code exists for "material recovery facilities (56292), this corresponds to establishments that handle MSW. Establishment of a new NAICS code for CDD materials recyclers would facilitate collection and aggregation of key materials and energy input data on a routine basis. NAICS was developed to be a dynamic industry classification and the classification is reviewed every 5 years to identify new or emerging industries. There is a proposal solicitation currently underway for new and merging industries for inclusion in the 2017 list, which presents an opportunity to include CDD recyclers with a unique NAICS code.

CDD Materials Transport

For the LCI presented in this report, a uniform transport distance of 20 km was assumed between the point of generation and the next step of management (e.g., processing facility, landfill). Different transport distances would impact the results of an LCA, but such detailed data are not currently available. This data gap could be addressed in several ways. First, the U.S. Census Bureau's commodity flow survey, which provides estimates of the distances various commodities are transported via different modes, represents an opportunity to include waste haulers, recyclers, and material processors in the future surveys to estimate the transport distances of CDD materials. The current commodity flow survey includes "waste and scrap," but details regarding the materials included and the universe of entities surveyed for the analysis are not available. A second option would be to conduct direct research of facilities by compiling average or typical transport distances for a variety of CDD management facility sizes (i.e., material quantity accepted) in different geographic areas (e.g., each of the 10 U.S. EPA regions).

Long-Term Liquid Emissions from Materials Deposited in Landfills

None of the U.S.-specific LCA models (WARM, MSW-DST) include liquid emissions from the disposal of materials at a CDD landfill. As discussed in the report, some MSW waste categories can be used as proxies for estimating liquid emissions from a few of CDD materials, and some European databases (e.g., Ecoinvent) include liquid emissions from CDD material disposal in a sanitary landfill. Although liquid emissions based on laboratory leaching data (SPLP and TCLP) for specific CDD components were presented in this analysis, this approach has limitations and additional research must be conducted to provide a more comprehensive and realistic view of liquids emissions at operating facilities. Example research areas include:

- Identifying realistic L:S ratios for establishing a leaching test framework to assess liquids emissions from a LCA perspective
- Including a larger listing of leached chemicals
- Including biological processes that result in leaching in addition to physico-chemical processes
- Including aggregated CDD materials rather than specific components, as the leaching behavior of a specific CDD component may differ in the presence of another CDD component (or, in like fashion, MSW components)
- The impact of the management of sludge from wastewater treatment plant used for treating leachate on the overall release of metals into the environment.

Long-Term Gaseous Emissions from Materials Deposited in Landfills

Gaseous emission estimates presented in this report only included methane, carbon dioxide, and (to a lesser extent) hydrogen sulfide and these estimates have several notable limitations. Methane generation potential (based on research conducted at North Carolina State University) for branches and OCC were used as a proxy to estimate the methane and biogenic carbon dioxide emissions for wood/LCD and gypsum drywall (paper fronting and backing), respectively. The researchers at North Carolina State University estimated methane generation potential of various MSW constituents based on bioassays conducted in 2-L reactors. These data have been widely used by various LCA models (e.g., WARM, MSW-DST). Although multiple studies investigated hydrogen sulfide emissions from drywall disposal in landfills, the hydrogen sulfide generation potential specifically as a result of drywall and RSM disposal in CDD landfills has not been reported. Hydrogen sulfide generation estimates have been the result of either laboratory testing or from the bulk disposal of MSW (which may contain other sulfur-containing materials). Measurements specific to CDD materials from larger-scale studies should be considered for future research to provide a better estimate of the emissions of major (i.e., carbon dioxide, methane) and minor (hydrogen sulfide, non-methane organic compounds) LFG constituents.

Several studies have documented how the cement in concrete can absorb atmospheric carbon dioxide over time in a process called *carbonation*. This mechanism of carbon dioxide uptake was not included in the LCI presented for concrete. Future studies should consider measurements of carbon dioxide uptake by concrete disposed of in landfills or used in other recycling applications.

Long-Term Performance of Recycled Material-Derived Products and Services

Although the use of recycled materials to replace primary resource extraction would generally reduce the overall impact on the environment, additional factors may reduce the anticipated benefits of recycling. For example, pavement made from recycled concrete aggregate and/or RAP may have a shorter service life compared to pavements manufactured entirely from primary materials. Additional research should attempt to quantify the serviceable life of materials manufactured from recycled materials on a per-mass-recycled basis and account for this lifespan difference in developing and updating LCI process datasets.

Capital Equipment and Land Development Burdens

Most LCI/LCA disregard emissions associated with the manufacturing of equipment and materials (e.g., steel, concrete used for facility construction) used for processing CDD materials. A common justification provided for this exclusion is that the environmental burdens associated with the manufacturing and production of products/equipment is generally inconsequential when compared to the impacts of the operations phase. Future research should consider assessing the impact of the exclusion of capital equipment burdens from LCA.

Furthermore, the process of greenfield development may disrupt naturally-occurring environmental services. While some LCA programs (e.g., Ecoinvent) and LCI datasets (e.g., Stripple 2001) attempt to quantify the environmental impact of land transformation and land-clearing activities (e.g., loss of carbon sequestration associated with biomass loss), there are other services that natural ecosystems provide that are more challenging to quantify (e.g., wetland treatment of stormwater runoff or process waters discharged from adjacent industry, the effect of noise cancellation provided by vegetation on undeveloped land between highways and neighborhoods). Without a consistent methodology to place a value on the environmental services provided by undeveloped land, realistic environmental burdens associated with land development are difficult to allocate. Additional research efforts should attempt to analyze and quantify the average environmental services provided by undeveloped land in the U.S.

Decommissioning and Disposal Burdens

Similar to capital equipment burdens, a majority of LCI do not quantify the impacts of facility/equipment decommissioning/disposal for the same reason described above. However, the manner in which a process-dedicated piece of equipment is managed at the EOL may have a significant impact on the overall emissions associated with that process. For example, if all the steel recovered from landfill operations equipment (e.g. compactors, excavators, dozers) was recycled for the production of new landfill operations equipment, the capital equipment burdens associated with virgin iron ore extraction and smelting would be avoided.

Operation and Maintenance Consumable Burdens

While it is likely that the bulk of emissions resulting from the operation of a particular process would occur as a result of energy use, almost all equipment requires the replacement of various fluids, filters, and worn mechanical components over the course of its service life. The environmental burdens resulting from the production of these consumable materials should be accounted for during the future development of LCI; until these emissions are quantified, it is not possible to estimate their impact on the overall emissions associated with that particular process.

11.3 References

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