Cradle-to-Grave Environmental Impacts of Mass Timber Construction in the United States: A Comparative Assessment to Conventional Construction Materials

Christina Bjarvin

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Reading Committee:

Indroneil Ganguly, Chair

Francesca Pierobon

Maureen Puettmann

Program Authorized to Offer Degree:

School of Environmental and Forest Sciences

University of Washington

Abstract

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Christina Bjarvin

Chair of the Supervisory Committee:

Indroneil Ganguly

School of Environmental and Forest Sciences

The use and end-of-life (EOL) phases of mass timber products (MTP)—including cross-laminated and glue-laminated timber—remain comparatively understudied in life cycle assessment (LCA) literature, leaving uncertainty around their cradle-to-grave environmental performance relative to conventional construction materials. This dissertation addresses that gap by building on assumptions used in prior U.S.-based LCA studies of MTP and other structural materials.

A comprehensive environmental assessment was conducted for two case-study buildings: Google's "MT1" mass timber building and an alternative reinforced concrete design. Material-specific impacts were evaluated across all life cycle phases—production, construction, use, and EOL—along with carbon storage benefits and product substitution benefits associated with reuse, recycling, and energy recovery. Primary and secondary data

were used to quantify material and energy inputs, and multiple EOL scenarios were modeled, including reuse, recycling, incineration, and landfill. Cradle-to-grave impacts were then compared under two waste management pathways: a Business-as-Usual (BAU) demolition scenario with minimal wood recovery, and an Optimistic deconstruction scenario with recovery of MTP for reuse.

Mass timber construction showed consistently lower global warming, acidification, eutrophication, smog formation, and ozone depletion impacts (-10% to -50%) than concrete in the production, construction, and use phases, but higher impacts during EOL treatment when MTP was not recovered (11% to 900%). Reuse of MTP in the Optimistic scenario reduced impacts in most categories; although diesel use during waste processing increased acidification and smog formation, these increases were more than offset by substitution benefits, yielding net reductions across all impact categories relative to landfilling. Accounting for temporary biogenic carbon storage using a dynamic radiative forcing framework—rather than applying conventional ISO carbon accounting rules, which credits higher storage to landfilling—further highlighted the climate benefits of reused MTP. Under the BAU scenario, MT1 outperformed the concrete building (-8% to -61%) on a cradle-to-grave scope in all categories except eutrophication; when MTP was reused, eutrophication impacts became comparable between the two designs.

Overall, the findings demonstrate that mass timber construction offers clear cradle-to-grave environmental advantages over reinforced concrete, particularly when structural MTP is recovered for reuse. While landfill disposal can obscure these benefits under ISO carbon accounting, circular economy pathways—reuse, reprocessing, recycling, and energy recovery—consistently reduce environmental burdens and improve climate outcomes. The results underscore the decisive influence of carbon accounting methods and the need for frameworks that recognize the climate value of temporary biogenic carbon storage.

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Abbreviations

AP	Acidification Potential
BOD	Biological Oxygen Demand
CA	California
CO_2e	Carbon Dioxide Equivalent
COD	Chemical Oxygen Demand
BCRP	Biogenic Carbon Removal from Product
BCEP	Biogenic Carbon Emission from Product
BF	Board Foot
C&D	Construction and Demolition
CFC	Chlorofluorocarbons
CFR	Code of Federal Regulations
CLT	Cross-laminated Timber
DRI	Direct Reduced Iron
EAF	Electric Arc Furnace
EOL	End-of-Life
EP	Eutrophication Potential
EPA	Environmental Protection Agency
EPD	Environmental Product Declaration
GHG	Greenhouse Gas
GIS	Geographic Information System
GLT	Glued-laminated Timber
$GWP_{BIOGENIC}$	Global Warming Potential from Biogenic Sources
GWP_{FOSSIL}	Global Warming Potential from Fossil Sources
GWP_{TOTAL}	Global Warming Potential from Biogenic and Fossil Sources

INW	Inland Northwest
ISO	International Organization for Standardization
kg	Kilogram
kW	Kilowatt
kWh	Kilowatt-hour
L	Liter
LA	Los Angeles
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LFG	Landfill Gas
m^2	Square Meters
m^3	Cubic Meters
MDF	Medium-Density Fiberboard
MF	Melamine Formaldehyde
MJ	Megajoule
MRF	Material Recovery Facility
MSW	Municipal Solid Waste
MTP	Mass Timber Panels
MUF	Melamine Urea Formaldehyde
NE	Northeast
NE-NC	Northeast-Northcentral
ODP	Ozone Depletion Potential
OSHA	Occupational Safety and Health Administration
PEP	Polymer Emulsion Polyurethane
PNW	Pacific Northwest
PRF	Phenol Resorcinol Formaldehyde
PUR	Polyurethane
RCI	Recycling Certification Institute
RIC	Reuse Innovation Center
SE	Southeast
SFP	Smog Formation Potential
SI	Substitution Impact
SLW	King County Salvage Lumber Warehouse
tkm	Tonne-kilometer
TRACI	Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts
U.S.	United States of America
VOC	Volatile Organic Compound
WARM	Waste Reduction Model

Overarching Introduction

Background

Human activities and the greenhouse gas (GHG) emissions they generate have significantly altered the natural systems that support human life. The scale and speed of these changes have pushed the planet into the Anthropocene, a geological epoch defined by rapid, human driven climate and ecosystem disruption. As global systems approach irreversible tipping points, reducing total GHG emissions—also referred to as embodied carbon—has become increasingly important.

The built environment represents a major opportunity for emission reductions. Buildings and construction account for 39% of global GHG emissions, and cement production alone contributes roughly 8%, driven by the global annual production of 4 billion tons of cement (Abergel et al., 2018). Reducing the reliance on conventional, high-embodied-carbon materials such as concrete is therefore a key climate mitigation strategy.

Life cycle assessment (LCA) provides a standardized framework for evaluating the environmental impacts of building materials across their full life cycle. The four stages of LCA—goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation—enable researchers to quantify material and energy inputs, emissions, and resulting environmental impacts. Depending on the system boundary, LCAs may assess impacts from material production alone (cradle-to-gate), from production through construction (cradle-to-site), or across the full life cycle including end-of-life (cradle-to-grave). LCIA results typically include impact categories such as global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), ozone depletion potential (ODP), and smog formation potential (SFP).

Literature Review

LCA research has demonstrated that material choices in construction can substantially reduce embodied carbon (Duan et al., 2022; Puettmann et al., 2021; Hemmati et al., 2024). Mass timber panels (MTP)—including cross-laminated timber (CLT) and glue-laminated timber (GLT)—have emerged as lower-carbon alternatives to reinforced concrete in mid- to high-rise buildings. A global review of whole-building LCA (WBLCA) studies found that substituting MTP for reinforced concrete can reduce cradle-to-gate embodied carbon by an average of 43% (Duan et al., 2022). North American studies report similar findings: comparative assessments of 8-, 12-, and 18-story buildings show 22–50% reductions in cradle-to-site embodied carbon (Puettmann et al., 2021), and a five-story residential building in Arkansas showed a 19% reduction when MTP replaced a steel-and-concrete structure (Hemmati et al., 2024).

Comparative life cycle assessment (LCA) studies of mass timber panels (MTP) and reinforced concrete systems have been conducted across a range of system boundaries, from cradle-to-site (A1–A5) to cradle-to-grave (A1–C4). Most existing research focuses on cradle-to-site impacts, where substituting reinforced concrete with MTP in structural and non-structural elements has been shown to reduce embodied carbon by an average of 43% (Duan et al., 2022). Although fewer studies extend to the full cradle-to-grave scope due to uncertainties in end-of-life (EOL) processes, the available literature similarly finds lower embodied carbon for MTP buildings when use-phase and EOL impacts are included (Chen et al., 2020; Liang et al., 2021).

Of the 45 assessed studies in Duan et al.'s (2022) global review, 57% adopted a cradle-to-grave scope. However, only two of these studies were conducted in the United States (U.S.) (Chen et al., 2020; Liang et al., 2021). Since then, only a handful of North American WBLCA studies have been added to existing literature (Kumar et al., 2024; Risha et al., 2025). Limitations in these studies highlight persistent gaps in understanding the comparative whole-life performance of U.S.-produced mass timber and concrete buildings. Key modules in the use phase and EOL phase—particularly waste processing—were either poorly documented or excluded due to a lack of data, restricting reproducibility. Similar gaps in the use and EOL phases exist for WBLCA studies in Europe, Asia, and Oceania (Vidal et al., 2019; Liu et al., 2016; Jayalath et al., 2020; Lu et al., 2017).

In the use phase, North American MTP WBLCA studies commonly exclude the impacts of MTP use (B1) and repair (B3) because of the limited availability of data specific to MTP construction (Liang et al., 2021; Chen et al., 2020; Kumar et al., 2024; Risha et al., 2025). Many of these same studies include maintenance (B2) and replacement (B4) impacts, yet provide little documentation of the modeling approaches used for these modules. One study offers a notable exception: Andersen et al. (2022) provide comprehensive documentation of their use-phase methods, applying material lifespan modeling in which the average service life of each material determines the number of replacement cycles. When material service lives are used to model maintenance and repair, non-MTP WBLCA studies report a wide variation in impacts, driven largely by differences in methodological choices and practitioner assumptions (Francart et al., 2021). Together, these findings underscore the need for more transparent reporting of use-phase modeling in MTP WBLCA studies, which remains a persistent gap in the literature.

In the EOL phase, exclusions of life cycle modules in MTP WBLCA studies are less common than in earlier stages, but they are typically not well documented in the underlying life cycle inventory. Most North American studies relied on the Athena Impact Estimator for Buildings (IE4B), an open-source WBLCA tool (Chen et al., 2020; Kumar et al., 2024; Risha et al., 2025). IE4B provides LCI data for all four EOL modules—building demolition (C1), waste transportation (C2), waste processing (C3), and waste disposal (C4)—although it does not include the alternative C1 scenario of deconstruction. Its demolition methodology is based on two case study buildings (Athena Sustainable Materials Institute, 1997), while landfilling emission factors for wood are sourced from the U.S. Environmental Protection Agency's WARM tool (EPA, 2023a–b). WARM documents methods for modules C2–C4 across a range of construction waste types, but waste processing for recycling is excluded from both WARM and IE4B, highlighting a key data gap for a critical circular-economy strategy for waste wood.

The final North American MTP WBLCA study provided the most comprehensive methodological documentation among the four studies, yet it excluded modules C1 and C3 due to data limitations, and the distances used for waste transportation were not justified (Liang et al., 2021). That study supplemented IE4B with the DATASMART LCI database, which offers alternate waste disposal data for multiple waste types across landfill and incineration scenarios (LTS, 2023). However, DATASMART's disposal data are based on European landfill and incineration technology from 2000, which may not reflect current U.S. technology, regulatory requirements, or disposal-related emissions. Overall, existing MTP WBLCA studies do include EOL impacts, but limited transparency in methodological documentation continues to constrain replicability and hinder the assessment of broader environmental impacts in future studies.

In addition to the use and EOL phases, further gaps in North American MTP WBLCA studies arise in Module D, which accounts for environmental benefits and burdens from recovered products outside the system boundary. Reuse, recycling, and incineration with energy recovery can displace virgin material production or energy generation, and these avoided impacts are reported as substitution benefits under Module D (ISO, 2017).

The four North American MTP WBLCA studies primarily relied on the IE4B tool for Module D calculations. IE4B's background methodology details substitution benefits for recycled metals, but it does not specify whether substitution benefits for recycled wood are included. Three studies used IE4B's default assumptions to calculate Module D impacts (Chen et al., 2020; Kumar et al., 2024; Risha et al., 2025), while the fourth calculated substitution benefits independently (Liang et al., 2021). That study assumed that 70% of MTP would be recovered from the landfill, with 52.5% reused and 17.5% incinerated. Reused MTP was assumed to displace virgin MTP production, and incinerated MTP was assumed to displace natural gas at 77% efficiency. However, all four studies also reported permanent biogenic carbon storage under Module D using IE4B's default assumptions.

IE4B applies the +1/–1 biogenic carbon accounting method consistent with ISO 21930:2017, in which biogenic carbon removals and storage are expressed as –1 kg CO_2e per kg CO_2, and biogenic carbon emissions are expressed as +1 kg CO_2e per kg CO_2 (ISO, 2017). Emissions may occur to the atmosphere (e.g., incineration or landfilling) or outside the system boundary (e.g., product recovery through reuse or recycling). Under this method, 100% of biogenic carbon is emitted for incineration, reuse, or recycling, whereas landfilling assumes that 90% of biogenic carbon remains permanently stored after decomposition, resulting in a net negative cradle-to-grave storage credit (ISO, 2025). Consequently, the cradle-to-grave biogenic carbon emissions reported for the four North American case study buildings are negative, reflecting IE4B's default assumption that 27.4% of wood is recycled and 72.6% is landfilled (Athena Sustainable Materials Institute, 2023). If all MTP waste were instead recovered for reuse or recycling, net emissions would be neutral, illustrating how current ISO guidelines create a carbon-storage incentive for landfilling.

Alternative approaches have been developed to account for temporary rather than permanent carbon storage, including ton-year methods such as the Moura-Costa and Lashof approaches, as well as dynamic LCA (Moura-Costa & Wilson, 2000; Fearnside et al., 2000; Levasseur et al., 2010). Ton-year methods assign credits based on the duration of carbon storage, using equivalency factors to convert ton-years of storage into avoided CO_2 emissions. Dynamic LCA instead uses time-dependent life cycle inventories and characterization factors to account for the timing of emissions.

Comparisons of ton-year approaches and dynamic LCA have shown that dynamic LCA offers greater flexibility for afforestation and land-use change projects because it captures emissions timing across life cycle stages and is not restricted to a static 100-year time horizon (Levasseur et al., 2012). However, no study has yet compared temporary storage benefits calculated through dynamic LCA with permanent storage benefits calculated under ISO guidelines across multiple product systems.

Taken together, the lack of existing data and the low transparency in methods across the use phase, EOL phase, and Module D benefits beyond the system boundary remain persistent limitations in current North American MTP WBLCA studies. These gaps make it difficult to compare or validate results and hinder the ability to assess the environmental trade-offs among different waste treatment scenarios for MTP—such as reuse, recycling, incineration with energy recovery, or landfilling—which can lead to substantially different impact profiles. This underscores the need for a more detailed and transparent evaluation of each life cycle phase to support a complete cradle-to-grave WBLCA for mass timber buildings in the U.S.

Research Objectives

The overarching goal of the research is to assess the environmental impacts and benefits associated with MTP construction from cradle to grave in comparison to reinforced concrete construction. Beyond academic applications, the results have practical value for policymakers, LCA practitioners, and material specifiers. States such as Washington and California are advancing Buy Clean/Buy Fair legislation and embodied-carbon-based procurement requirements, and more jurisdictions are beginning to rely on whole-building LCA in building codes and public project evaluations. By supplying transparent, U.S.-specific LCA data for MTP and other C&D materials, this research can support more accurate policy implementation, inform material purchasing preferences, and help fill critical data gaps in procurement frameworks and state-level embodied carbon standards.

Addressing the data gaps identified in existing MTP LCA research is essential for improving understanding of MTP's environmental impacts across its full life cycle. This dissertation aims to fill these gaps with a comprehensive analysis of MTP and other C&D materials in each life cycle module. Each chapter is dedicated to a different life cycle phase:

- **Chapter 1**: Production, Construction, Use
- **Chapter 2**: End-of-Life
- **Chapter 3**: Additional Environmental Benefits
- **Chapter 4**: Cradle-to-Grave

The fourth chapter will synthesize the findings of the previous three chapters into a full cradle-to-grave analysis for two case study buildings. Two case study buildings were selected for this analysis: Google's "MT1" building and its alternative concrete design ("Concrete Baseline"). Descriptions of each building are provided in Chapter 1.

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Chapter 1: Environmental Impacts of the Production, Construction, and Use Phases

Introduction

Comparative life cycle assessment (LCA) studies of mass timber panels (MTP) and reinforced concrete systems have been conducted across a range of system boundaries, from cradle-to-site (A1–A5) to cradle-to-grave (A1–C4). Most existing research emphasizes cradle-to-site impacts, where substituting reinforced concrete with MTP in structural and nonstructural elements has been shown to reduce embodied carbon by an average of 43% (Duan et al., 2022). However, Duan et al.'s (2022) global review also highlights that relatively few studies consider material maintenance and repair during the use phase.

Compared with the well-documented production and construction phases (A1–A5), portions of the use phase (B1–B5) remain a notable gap in the literature on mass timber buildings. This gap reflects the relatively recent adoption of MTP in mid- to high-rise construction (Balasbaneh and Sher, 2022). The use phase comprises seven modules—use (B1), maintenance (B2), repair (B3), replacement (B4), refurbishment (B5), operational energy use (B6), and operational water use (B7)—each of which may contribute to long-term environmental impacts.

While emissions during material use (B1) typically do not occur for construction materials (Athena Sustainable Materials Institute, 2023), maintenance (B2) may be required to preserve structural performance. For example, debris accumulation on uncoated steel joints can lead to corrosion; concrete can shrink or crack under moisture exposure; and reinforcing materials may deteriorate through carbonation or corrosion (Sarma & Adeli, 2002; Rudbeck, 1999). Wood components are similarly vulnerable to moisture-related deterioration, though proper installation and the use of preservatives can mitigate these risks (Rudbeck, 1999). Repair (B3) is typically excluded from WBLCA studies, due to data gaps and inherent uncertainty in predicting repairs needed for damage incurred to materials (Athena Sustainable Materials Institute, 2023).

Interviews with architects at atelierjones, a Seattle-based firm, underscored the importance of design and installation practices that minimize future maintenance needs—such as selecting surface-treated lumber or designing assemblies that allow ventilation behind façades. Yet widespread knowledge gaps persist among designers and contractors regarding proper storage, installation, and maintenance of MTP. These gaps are partly attributable to the fact that MTP construction was only incorporated into the International Building Code in 2015 (Zelinka et al., 2020).

When maintenance and repair are insufficient, components may require full replacement (B4). ISO defines service life as the period during which a component meets performance requirements, and ISO 15868:2011 recommends that inaccessible or difficult-to-replace components match the building's design life, while major replaceable components in a 60-year building should have a 40-year design life (ISO, 2011). Refurbishment (B5)—including retrofits and renovations—introduces new materials to improve energy performance, aesthetics, or spatial design (Vilches et al., 2017).

Research Objectives

Addressing the data gaps identified in existing MTP LCA use-phase research is essential for improving understanding of MTP's environmental impacts across its full life cycle and its comparative performance relative to other C&D materials. This chapter evaluates the production, construction, and use-phase environmental impacts associated with a case-study MTP building and an alternative concrete design. This is achieved by compiling existing data, validating assumptions through literature and surveys, and comparing environmental impacts by building component and material type.

The research objectives are to:

1. **Evaluate the bill of materials** for two case-study buildings to establish a consistent basis for comparing material quantities and component-level impacts.
2. **Compile production-phase (A1–A3) data** for each material identified in the bill of materials, drawing from existing LCA datasets and literature to ensure consistency and transparency.
3. **Apply fuel-consumption models** to quantify the environmental impacts associated with building construction activities, including equipment use and on-site processes.
4. **Compile secondary data and assumptions for use-phase (B1-B4) activities—** maintenance, repair, and replacement—for enclosure and interior materials, integrating literature, survey responses, and industry guidance.

The overarching goal of this chapter is to assess the environmental impacts associated with the production, construction, and use of C&D materials, thereby improving understanding of the environmental trade-offs and benefits associated with each material when used in different building components. This research provides foundational data that can strengthen future LCA studies incorporating the use phase for MTP products or buildings, and it can be adapted to different building designs.

Methods

Goal & Scope

This LCA was conducted in accordance with ISO 14040, ISO 14044, and ISO 21930 (ISO, 2006a-b; ISO, 2017). All impacts were modeled in SimaPro v9.5 using the TRACI characterization method (Bare, 2012). The functional unit for this analysis is one building.

The study includes all modules of the production (A1-A3), construction (A4-A5), and use (B1-B4) phases (Figure 1). Figure 2 illustrates the system boundary and shows how processes and materials are included in each life cycle phase.

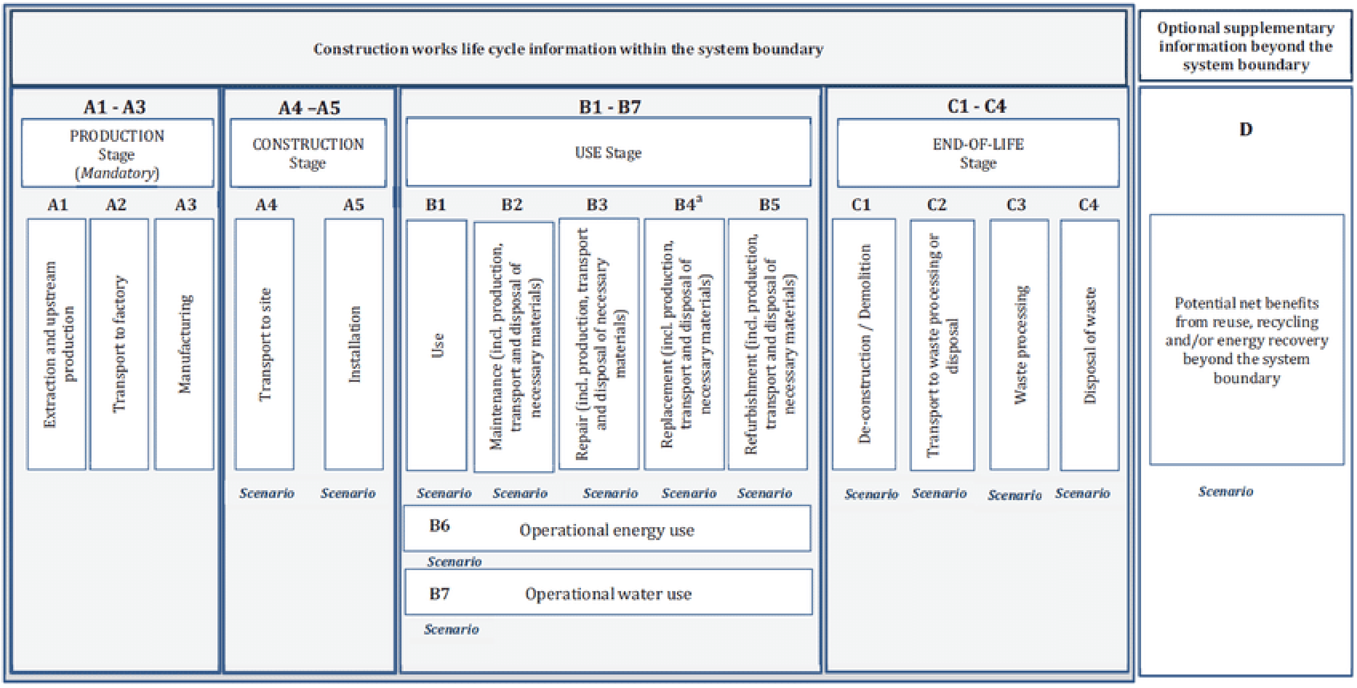

Figure 1. Life cycle phase diagram for life cycle assessment of building products (ISO, 2017).

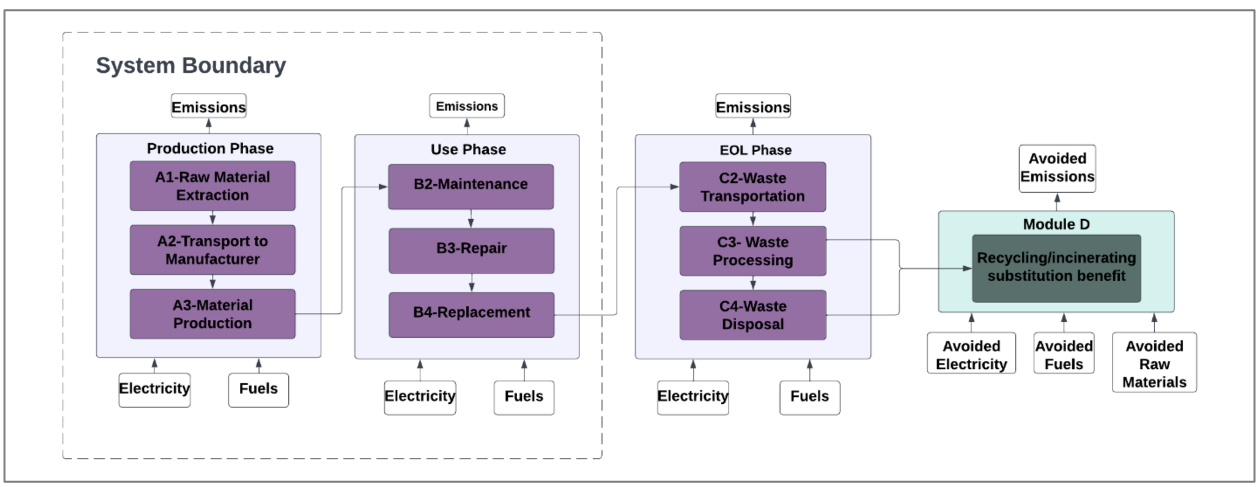

Figure 2. System boundary diagram for the production, construction, and use phases of construction and demolition waste. Modules outside the dashed line are not included in the system boundary.

This study evaluates the five mandatory impact categories specified in ISO 21930:2017 (

Table 1).

Table 1. Impact category indicators per ISO 21930 (ISO, 2017).

Impact Categories	Abbreviation	Units	Method
Global warming potential, fossil	GWP_{FOSSIL}	kg CO_2e	TRACI 2.1 V1.08
Acidification potential	AP	kg SO_2e	TRACI 2.1 V1.08
Eutrophication potential	EP	kg Ne	TRACI 2.1 V1.08
Ozone depletion potential	ODP	kg $CFC\text{-}_{11}e$	TRACI 2.1 V1.08
Smog formation potential	SFP	kg O_3e	TRACI 2.1 V1.08

Case Study Buildings

The case study buildings selected for this dissertation are Google's MT1 building and its alternative concrete design ("Concrete Baseline"). MT1 is a five-story commercial building that was completed in 2021, located at 1265 Borregas Avenue, Sunnyvale, California (CA) (Figure 3-Figure 4). The building contains 3,100 m^3 of CLT and 2,000 m^3 of GLT across its structure and enclosure. Structural MTP elements include CLT floors and walls, and glulam columns, beams, girders, and purlins. Enclosure MTP is used in the roofing system. The building has a floor area of 182,500 ft^2 (16,955 m^2) and a height of 31 ft (9.45 m).

Figure 3. Google's MT1 building at 1265 Borregas, Sunnyvale, California. Photo by Christina Bjarvin.

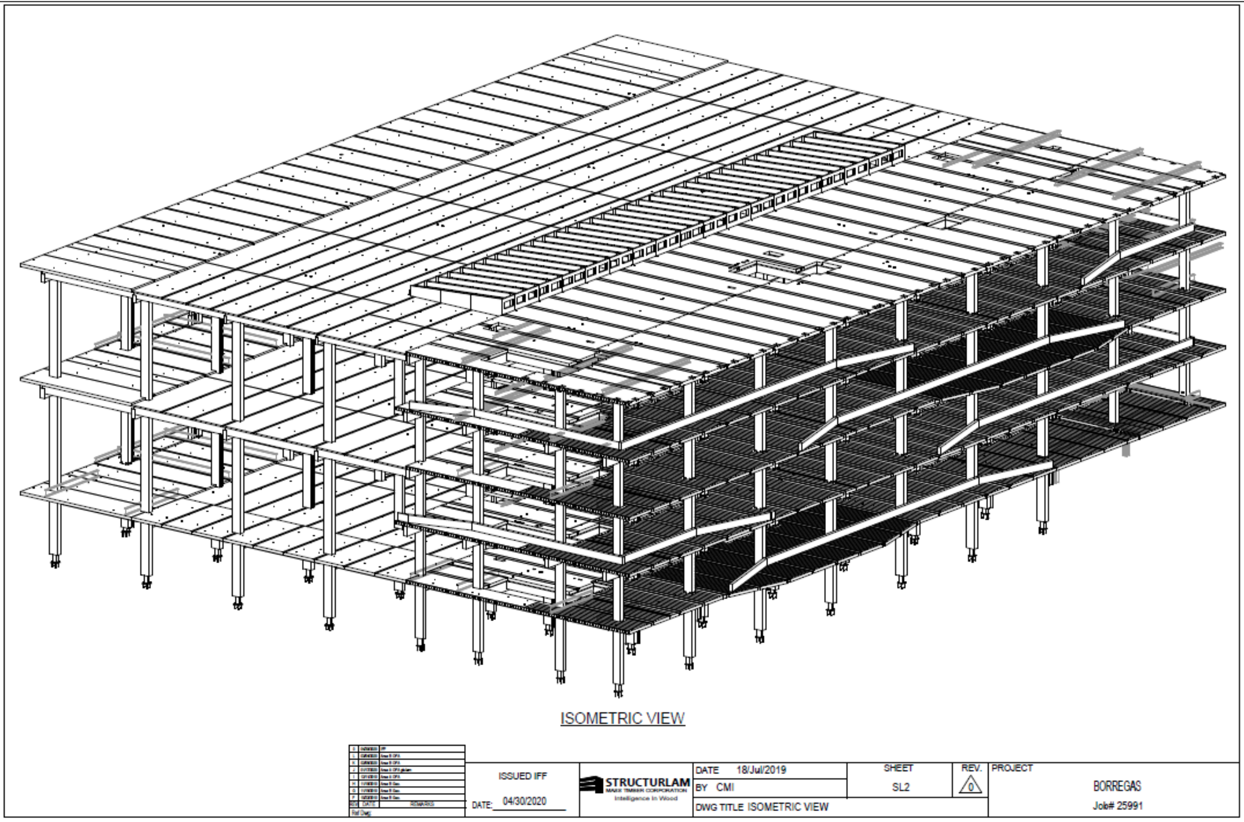

Figure 4. Elevation view drawing of Google's MT1 building.

A comprehensive bill of materials for MT1 and the Concrete Baseline was developed by Atelier Ten, the environmental design consultants for the project. The bill of materials specifies the quantities of materials used in each building component—including the foundation, structure, enclosure, and interior—reported in square feet (ft^2), cubic feet (ft^3), or kilograms (kg). All materials were converted to kilograms using density, surface weight, or thickness values.

When available, product information sheets or manufacturer-specific EPDs were used to determine these metrics. If the bill of materials did not identify a manufacturer, or if manufacturer data were unavailable, values were sourced from industry-average EPDs or EPDs for similar products. The total mass of each material used in both buildings is summarized in the bill of materials in **Error! Reference source not found.**Table 2. Table 3 summarizes the material classifications for both buildings in different building components

Table 2. Bill of materials in the MT1 and Concrete Baseline case study buildings.

Material	MT1 Mass (kg)	Concrete Baseline Mass (kg)
Aluminum	36,328	46,119
Concrete	7,042,348	11,822,830
Gypsum	273,902	266,502
Inert	1,082,678	1,954,439
MTP	2,698,433	0
Paint	3,696	3,696

Steel	1,909,884	3,439,371
Wood	97,855	118,228
Total	*13,145,126*	*17,651,184*

Table 3. Classification of materials by component for MT1 and the Concrete Baseline.

Component	MT1 Materials	Concrete Baseline Materials
Foundation	Reinforced concrete	Reinforced concrete
Structure	Reinforced concrete, steel, CLT, GLT, lumber	Reinforced concrete, steel, gypsum
Enclosure	Concrete, glass, gypsum, insulation, roofing membranes, aluminum, steel, CLT, GLT, lumber, plywood, wood cladding	Concrete, glass, insulation, roofing membranes, aluminum, steel, lumber, wood siding
Interior	Concrete, carpet, tile, gypsum, insulation, paint, steel, lumber, plywood	Concrete, carpet, tile, gypsum, insulation, paint, steel, lumber, plywood

Production & Construction (A1-A5)

Production-phase (A1–A3) impacts for all C&D materials in the case study buildings were compiled through a structured review of EPDs and LCI databases. A consistent hierarchy was applied to identify the most representative production data for each material:

1. Manufacturer-specific EPDs (when the manufacturer was identified in the bill of materials)
2. U.S. industry-average EPDs
3. DATASMART datasets (LTS, 2023)
4. EPDs for similar products from other manufacturers

When an LCI source reported impacts in a functional unit other than kilograms, values were converted to a per-kilogram basis using product density, surface weight, or thickness information provided in the EPD or manufacturer documentation.

Each building's bill of materials was then modeled in SimaPro by matching material types and quantities to the corresponding A1–A3 datasets, enabling a contribution analysis of each component's share of whole-building impacts.

Construction-related impacts were evaluated next. Impacts of transportation to the construction site (A4) were calculated using a ton-kilometer method, multiplying the total mass of materials (in metric tons) by the transport distance (in kilometers). Transport distances were sourced from Atelier Ten's LCA report, and material masses were derived from the bill of materials. Construction (A5) impacts were estimated using Athena's crane diesel consumption model, which relates fuel use to the mass of materials and the average building height (Equation 1). This approach provided a consistent and transparent method for quantifying construction-phase emissions.

Equation 1. Diesel fuel consumption for lifting materials.

$$F = 0.000037 * M * H + \left(\frac{M}{500}\right) + 0.83,$$

Where:

- F = diesel consumption (L)
- M = mass of materials (kg)
- H = building height (m)

Use (B1-B4)

The use phase (B1–B4) encompasses use, maintenance, repair, and replacement of building materials. In this analysis, materials or assemblies with service lifespans shorter than those of the MT1 or Concrete Baseline buildings are assumed to undergo maintenance, repair, and/or replacement during the 75-year building lifespan. No emissions are assumed to occur during use (B1), while refurbishment (B5), operational energy use (B6), and operational water use (B7) are excluded.

A Swiss database reporting descriptive statistics for service lifespans of building subcomponents (Goulouti et al., 2021) was used to determine expected lifespans for foundation, structural, enclosure, and interior elements. XL Construction confirmed that these values are applicable to MT1. Reported median lifespans include:

- 78 years for foundation materials
- 74–94 years for structural materials (e.g., load-bearing walls, columns, beams)
- 14–50 years for enclosure materials (e.g., façades, coatings, windows, roof elements)
- 25–50 years for interior materials (e.g., internal walls, flooring, wall coverings

Given MT1's estimated 75-year lifespan, foundation and structural materials are assumed not to require maintenance, repair, or replacement. Enclosure and interior materials, however, are assumed to undergo these activities. Because the service lifespan database aggregates multiple materials within each building element, the reported lifespans are applied uniformly to all materials within a given subcomponent.

LCI data for maintenance, repair, and replacement were then compiled. The Athena Sustainable Materials Institute (Athena, 2002) provides B2–B4 inputs for enclosure assemblies such as cladding, siding, roofing, curtain walls, and window systems. Enclosure materials in MT1 and the Concrete Baseline were matched to the most appropriate Athena assemblies. Athena also reports the average service lifespan of each assembly. The number of cycles required over the 75-year building lifespan was calculated by dividing the building lifespan by the assembly lifespan and rounding to the nearest whole number (

Table 4).

Table 4. Classification of materials used in the enclosure and interior of MT1 and the Concrete Baseline, along with service lifespans and replacement/maintenance cycles for each material over the buildings' 75-year life spans.

Assembly	Material	Service Lifespan (Years)	# of Cycles

Enclosure			
Envelope	Rubber gaskets	35	2
Envelope	Insulation	35	2
Envelope	Aluminum extrusions, shading	25	2
Envelope	Glass	20	3
Envelope	Concrete	50	1
Envelope	Steel cladding or siding	50	1
Envelope	Wood cladding or siding	50	1
Envelope	Insulation	50	1
Roofing	Insulation	15	4
Roofing	Lumber	15	4
Roofing	Plywood	15	4
Roofing	Vapor barrier	15	4
Roofing	Steel	15	4
Roofing	TPO membrane	15	4
Interior			
Ceiling coverings	Acoustic ceiling panels	37	2
Wall coverings	Gypsum drywall, tile, paint	30	2
Finished flooring	Carpet, tile	25	2
Internal walls	Plywood, lumber, steel partition wall	50	1
Internal wall supports	Steel (studs, structural sections, galvanized sheets)	40	1

Athena's material inputs are reported per square meter of assembly for one maintenance, repair, or replacement cycle. These values were converted to a per-kilogram basis using product densities and surface weights from manufacturer documentation. Inputs per kilogram for one cycle were multiplied by the total number of cycles to obtain the total B1–B4 inputs per kilogram of each enclosure material; these values are reported in Appendix A.

Because Athena's B2–B4 data apply only to enclosure assemblies, a separate method was used for interior materials. Inputs for one replacement cycle were estimated based on service lifespans and national average waste management practices. Interior materials were classified according to the interior element categories in Goulouti et al. (2021), and the number of replacement cycles was calculated by dividing the building lifespan by the element's service lifespan. For each interior material, one replacement cycle consists of disposing 100% of the material to landfill[1] and producing an equivalent quantity of virgin material.

Results & Discussion

The LCIs for each module of the building life cycle were modeled in SimaPro v9.5 (LTS, 2023). Five impact categories were evaluated using the TRACI characterization method (Bare, 2012): fossil global warming potential (GWP_{FOSSIL}), ozone depletion potential (ODP), acidification potential (AP), eutrophication potential (EP), and smog formation potential (SFP). The LCIA results for each

[1] For concrete, steel, gypsum, and wood products, EPA (2018) waste distribution assumptions for recycling, incineration, and landfilling were applied.

module within the system boundary are presented in the following subsections, beginning with A1–A5 (production and construction).

Production & Construction (A1-A5)

A contribution analysis of production (A1–A3) impacts shows that the structure is the dominant assembly for both buildings, contributing an average of 41% across the five impact categories (Figure 5). This is followed by the interior (33%), enclosure (18%), and foundation (8%). The structure is the largest contributor in all categories except MT1's GWP_{FOSSIL} and both buildings' ODP, where interior materials dominate due to insulation-related emissions.

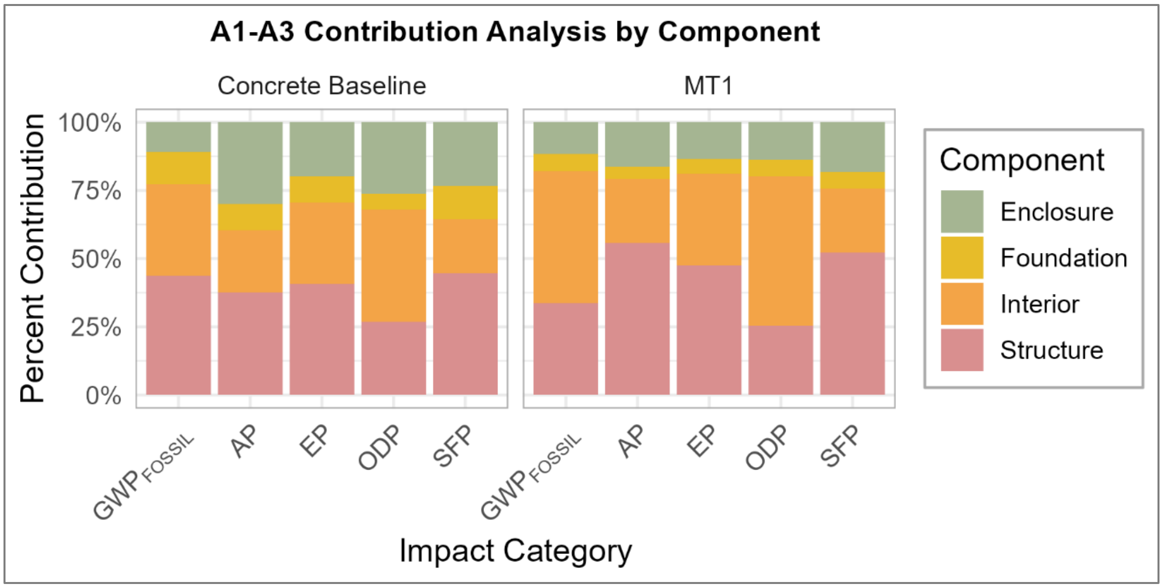

Figure 5. Contribution analysis of each building component to the production phase (A1-A3) environmental impacts of the MT1 and Concrete Baseline case study buildings.

A second contribution analysis by material type reveals that steel and other metals are the largest contributors in both buildings, with wood (including MTP and non-MTP wood) also contributing substantially in MT1 (Figure 6). Insulation drives the ODP impacts for both buildings. The Concrete Baseline uses 1.8 times more steel than MT1, while MT1 uses 26 times more wood than the Concrete Baseline. As a result, MT1 shows lower contributions from steel and metals but higher contributions from wood. The smaller difference in concrete contributions between the two buildings suggests that MTP primarily displaces steel emissions rather than concrete. The largest reductions in MT1 relative to the Concrete Baseline occur in GWP_{FOSSIL} and ODP, indicating that mass timber construction offers its strongest benefits in climate change and ozone depletion mitigation.

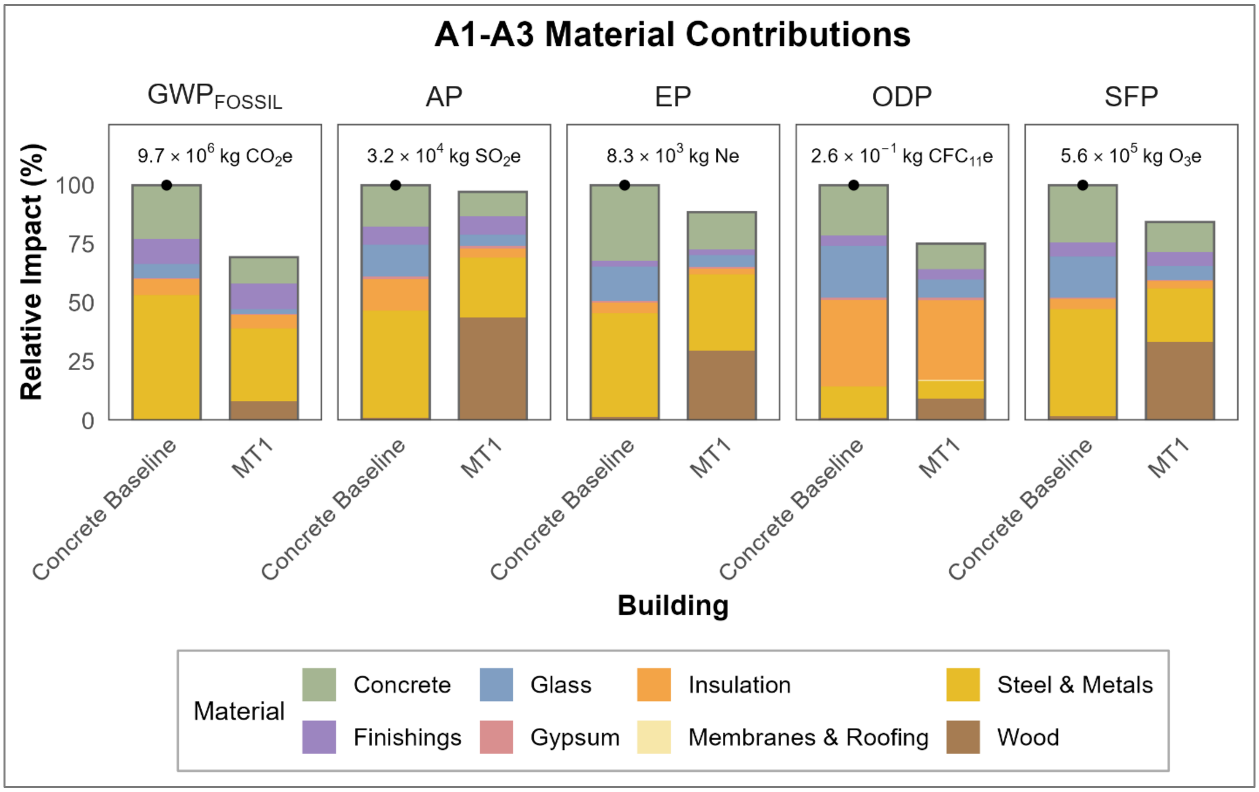

Figure 6. Relative impacts of the production phase (A1-A3) of the MT1 and Concrete Baseline case study buildings, with impacts normalized against the building with the largest impact in each impact category. The value of the larger impact of the two buildings is labeled for each impact category. The contributions of each material to the total A1-A3 impacts are also shown.

Across both buildings, production (A1–A3) impacts overwhelmingly dominate the total production and construction (A1–A5) impacts, contributing an average of 90% across all categories (Figure 7). A4 and A5 contribute 7% and 3%, respectively. In the ODP category, A4 and A5 contributions are negligible (<1%) due to the high ODP intensity of cement production. MT1 exhibits lower impacts than the Concrete Baseline in all categories, with an average reduction of 20% across the five impact categories.

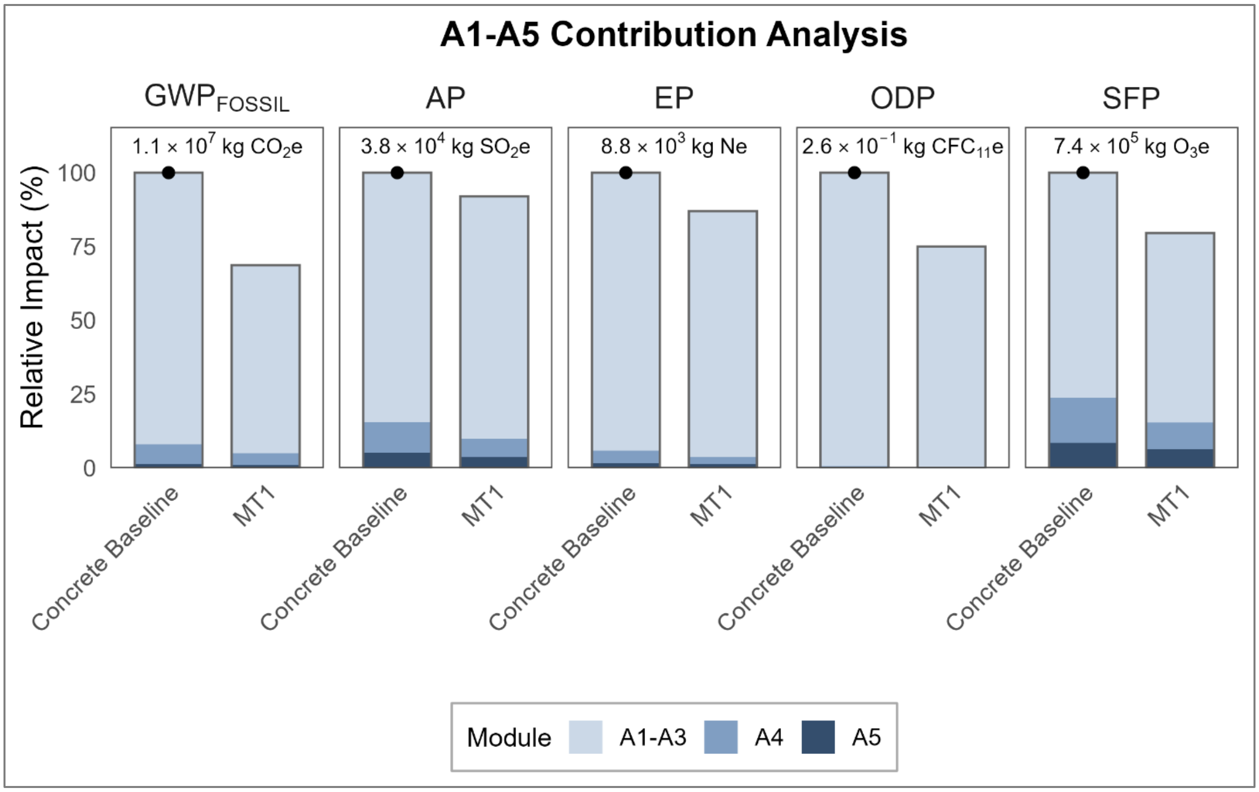

Figure 7. Relative impacts of material production and building construction (A1-A5) of the MT1 and Concrete Baseline case study buildings, with impacts normalized against the building with the largest impact in each impact category. The value of the larger impact of the two buildings is labeled for each impact category. The contributions of each life cycle module to the total A1-A5 impact are also shown.

To validate the SimaPro modeling approach, the A1–A3 GWP_{FOSSIL} results were compared to the values reported by Atelier Ten for both buildings (Table 5). For MT1, the SimaPro estimate was 7% higher than Atelier Ten's value, while for the Concrete Baseline it was 5% lower. These differences stem primarily from inconsistencies in the underlying LCA datasets: not all materials in the bill of materials had manufacturer-specific EPDs, and Atelier Ten did not always document the sources of their background data. This suggests that the modeling approach for A1-A3 impacts in this dissertation analysis is suitable for these case study buildings.

Table 5. A1-A3 GWP_{FOSSIL} impacts of the MT1 building and its alternate concrete design, estimated by this dissertation in SimaPro and compared to the impacts estimated by Atelier Ten.

Building Assembly	**MT1 GWP_{FOSSIL} (SimaPro)**	**MT1 GWP_{FOSSIL} (Atelier Ten)**	**GWP_{FOSSIL} % Change: MT1**	**Concrete GWP_{FOSSIL} (SimaPro)**	**Concrete GWP_{FOSSIL} (Atelier Ten)**	**GWP_{FOSSIL} % Change: Concrete**
Foundation	4.19E+05	3.73E+05	12%	1.13E+06	1.20E+06	-6%
Structure	2.27E+06	2.10E+06	8%	4.26E+06	4.38E+06	-3%
Enclosure	7.90E+05	8.89E+05	-11%	1.07E+06	1.20E+06	-11%
Total	*3.48E+06*	*3.36E+06*	*4%*	*6.47E+06*	*6.79E+06*	*-5%*

Use (B1-B4)

Emissions occur when materials and energy are consumed to maintain (B2), repair (B3), or replace (B4) building assemblies over the building's lifespan. In this LCA, emissions from the use module (B1) are assumed to be negligible. Within the B4 module, the impacts of disposing of a material (C3–C4) and replacing it with an equivalent amount of new material (A1–A3) are included. The impacts associated with producing additional virgin material drive the majority of the B1–B4 results, particularly for materials used in the enclosure and interior subcomponents of the case study buildings.

Figure 8 presents the relative use phase (B1–B4) impacts for both buildings, disaggregated into contributions from maintenance and repair (B2–B3) and replacement (B4). Values are normalized to the building with the higher impact in each category. In both buildings, most use phase impacts arise from replacement, reflecting the A1–A3 impacts of producing new materials during replacement cycles.

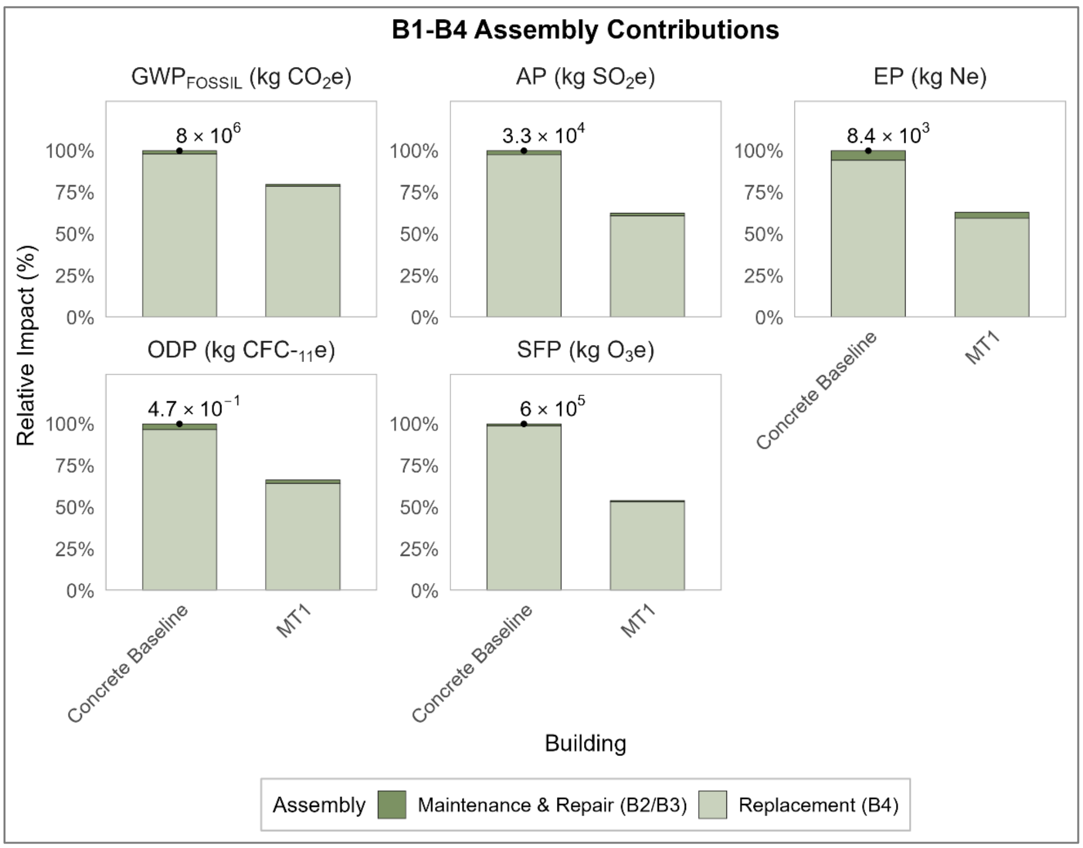

Figure 8. Relative impacts of use (B1-B4) for the MT1 and Concrete Baseline case study buildings, with contributions from each life cycle module. The value of the larger impact of the two buildings is labeled for each impact category.

Figure 9 shows the relative use phase impacts with contributions from different assemblies. MT1 had lower use phase impacts than the Concrete Baseline across all impact categories. Because the interior components are identical in both buildings, differences in use phase impacts stem entirely from maintenance, repair, and replacement of enclosure materials. To highlight these differences, the enclosure component is divided into its envelope and roof assemblies.

In both buildings, envelope materials contributed more to total use phase impacts than roofing materials. However, in MT1, interior assemblies had higher contributions than enclosure assemblies, while the opposite trend occurred in the Concrete Baseline. These trends are explored further through a material-level contribution analysis.

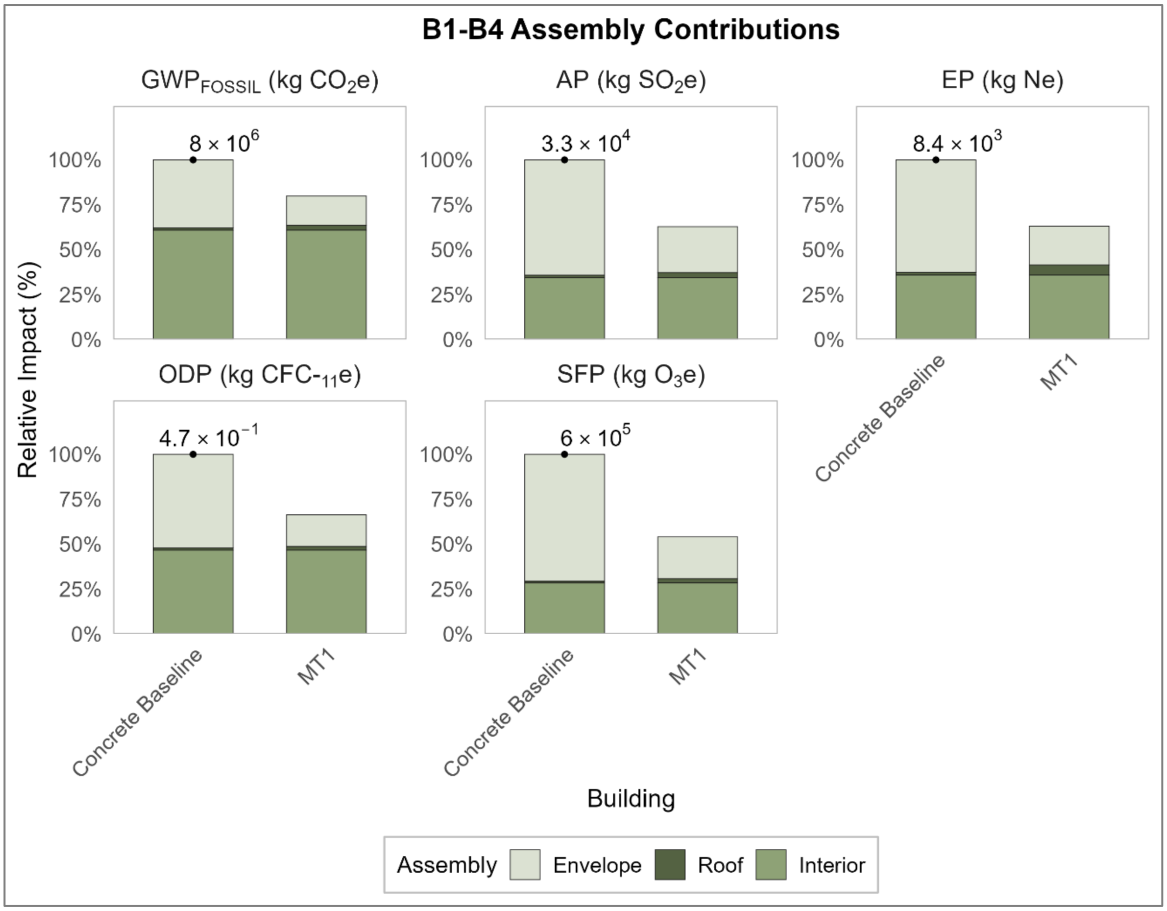

Figure 9. Relative impacts of use (B1-B4) for the MT1 and Concrete Baseline case study buildings, with contributions from the interior assembly and enclosure assemblies (exterior wall, roof) shown. The value of the larger impact of the two buildings is labeled for each impact category. Enclosure assemblies are represented by purple; the interior assemblies are represented by purple.

Figure 10 presents the relative use phase impacts with contributions from individual materials. The materials with the largest contributions vary by impact category and by building. In the Concrete Baseline, glass dominates across all impact categories. MT1 had minimal contributions from glass because it used 73 percent less glass in its envelope. Since glass is part of the exterior wall assembly, its replacement drives the large envelope contribution in the Concrete Baseline. Most of the use phase impacts for glass come from replacement (B4), as the A1–A3 impacts of producing new glass are much higher than the impacts of recaulking existing windows during maintenance.

In MT1, the materials with the largest contributions vary across impact categories. For GWP_{FOSSIL}, replacement of steel and interior finishes such as carpet, tile, and paint contributes substantially, driven by the fossil fuel consumption associated with manufacturing these materials. Steel replacement (B4) also contributes significantly to eutrophication potential (EP), due to phosphate emissions from slag disposal during steel production. Insulation replacement (B4) contributes to ozone depletion potential (ODP) because of upstream crude oil leachate emissions associated with electricity use during manufacturing.

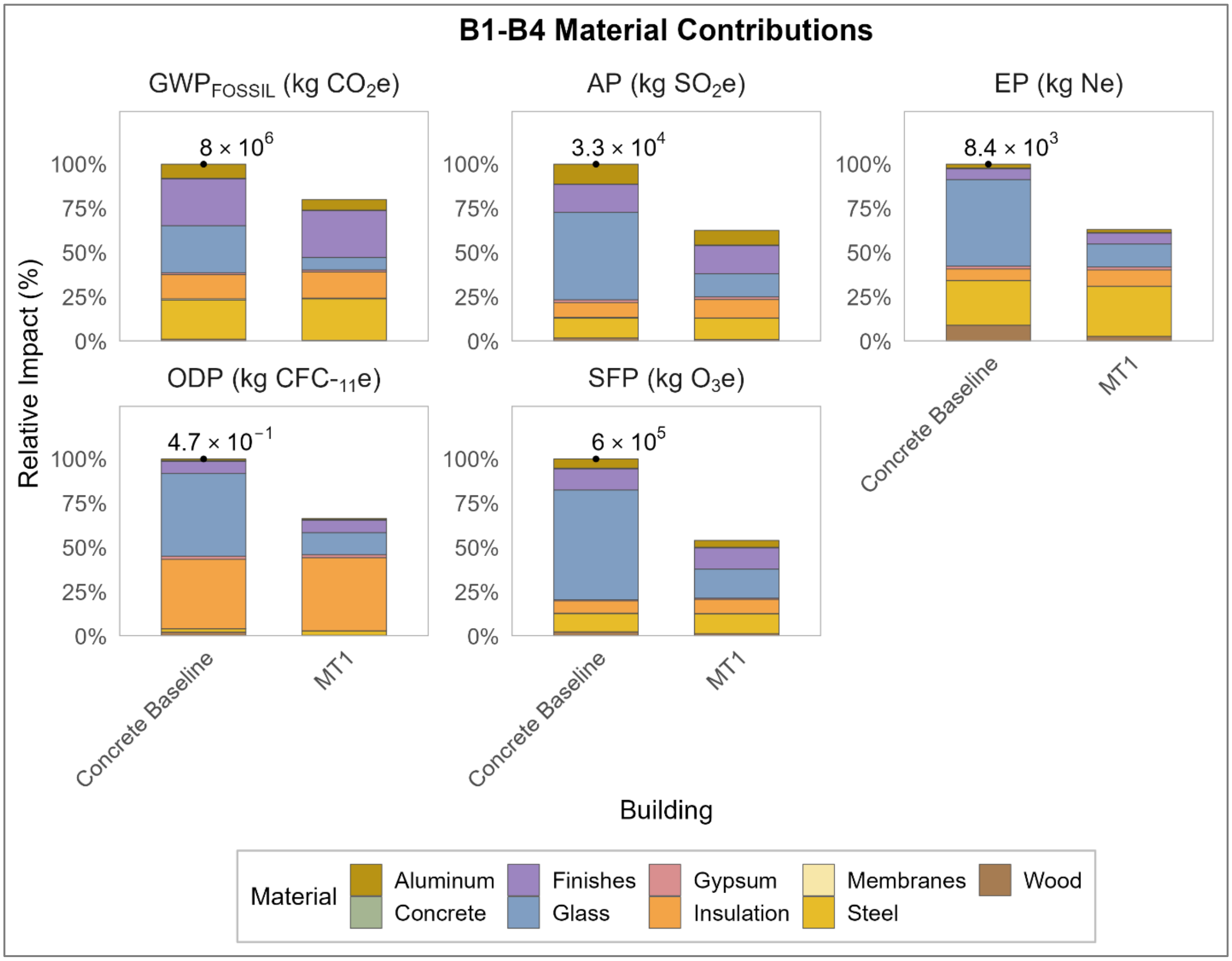

Figure 10. Relative impacts of use (B1-B4) for the MT1 and Concrete Baseline case study buildings, with contributions from individual materials shown. The value of the larger impact of the two buildings is labeled for each impact category.

Analyzing the results by assembly provides insight into how material selection and installation practices influence use phase impacts. Replacement (B4) accounts for most use phase impacts, with exceptions for materials requiring frequent maintenance or repair, such as wood siding, roofing membranes, or aluminum extrusions. Proper installation and maintenance can extend service lifespans, delaying or avoiding replacement cycles and reducing total use phase impacts.

Comparing assemblies and materials shows that glass replacement in the enclosure drives the higher use phase impacts in the Concrete Baseline relative to MT1, due to MT1's substantially lower glass use. In MT1, steel and finishing materials in the interior component account for most use phase impacts. Interior material replacement is often driven by occupant preferences and commonly occurs during tenant turnover. These impacts can be reduced by lengthening lease periods or encouraging occupants to maintain or retrofit interior materials rather than replacing them.

Limitations

Several limitations affect the interpretation of the production and construction (A1–A5) results. First, the production-phase (A1–A3) impacts rely on a hierarchy of data sources that includes manufacturer-specific EPDs, industry-average EPDs, and secondary LCI datasets. Not all materials in the bill of materials had manufacturer-specific EPDs, and in some cases, EPDs for similar products were used as proxies. These substitutions introduce uncertainty because manufacturing processes, energy mixes, and supply chains vary across producers.

Second, differences in functional units across EPDs and the units in the bill of materials required conversions using density, surface weight, or thickness values. These conversions assume uniform material properties, even though product specifications can vary by manufacturer or application. This may lead to over- or underestimation of impacts for materials with wide variability in composition or performance.

Finally, the modeling of construction impacts (A4–A5) depends on transport distances and crane fuel consumption estimates. Transport distances were sourced from a prior LCA report, but the underlying assumptions—such as supplier locations, routing, and vehicle types—were not documented. Similarly, the crane diesel consumption model provides a generalized relationship between material mass, building height, and fuel use, but does not account for construction sequencing, equipment efficiency, or site-specific logistics. The impacts of other construction equipment were not estimated in this model.

Several assumptions in the use phase (B1–B4) introduce uncertainty into the results. First, the analysis assumes that all structural materials require no maintenance, repair, or replacement over the 75-year building lifespan. This assumption holds only if design and installation practices prevent exposure to moisture or other degrading conditions. The same limitation applies to mass timber products used in the enclosure, where improper detailing or installation could shorten service life.

A second limitation arises from the use of Athena's maintenance and repair LCI inputs for enclosure materials (Athena, 2002). Because Athena reports use phase inputs per square meter of assembly, inputs for some materials were converted to a per-kilogram basis using material densities from two mass timber buildings that reported MTP use in their enclosures. These densities may not reflect the specific products or design requirements of other buildings, and applying them uniformly introduces potential inaccuracies. Additionally, the aggregated service lifespan categories in Goulouti et al. (2021) do not distinguish between individual materials within each building element, requiring uniform lifespans to be applied across diverse products.

Finally, interior material replacement was modeled using national average waste management practices and assumes that 100 percent of each material is disposed of and replaced with virgin material. In practice, reuse, selective replacement, or tenant-driven decisions could alter these flows. These assumptions simplify modeling but may not fully capture real-world variability in building operation and occupant behavior.

Conclusions

Across the production and construction phases (A1–A5), MT1 exhibited lower environmental impacts than the Concrete Baseline in all five impact categories. Contribution analyses showed that the structure was the dominant source of production impacts in both buildings, with MT1's mass timber components displacing substantial quantities of reinforced concrete and steel. Material-level results indicated that steel and other metals drove the Concrete Baseline's impacts, while MTP and other wood products contributed more to MT1's impacts. Production impacts far exceeded construction impacts in both buildings.

In the use phase (B1–B4), MT1 again had lower impacts across all categories. Because the interior assemblies were identical, differences arose entirely from enclosure materials. The Concrete Baseline used substantially more glass in its envelope, and glass replacement has high A1–A3 impacts, making it the dominant contributor to the Concrete Baseline's use phase impacts. In MT1, interior materials such as steel and finishes contributed more to use phase impacts, reflecting both their replacement cycles and the fossil fuel intensity of their production.

Replacement (B4) consistently contributed more to total use phase impacts than maintenance or repair (B2–B3), since replacement requires both disposal of old materials and production of new ones. As a result, the materials with the highest use phase impacts generally mirrored the trends observed in the A1–A3 results, with the number of replacement cycles further shaping total impacts.

Overall, MT1's lower impacts across both A1–A5 and B1–B4 reflect the combined effects of material substitution, reduced use of high-impact enclosure materials such as glass, and the lower production impacts associated with mass timber products.

Key Takeaways

This chapter evaluated the environmental impacts of the production (A1-A3), construction (A4-A5), and use (B1-B4) life cycle phases for a mass timber panel (MTP) case study building ("MT1") and its concrete alternative design. Several overarching findings emerged:

1. **Production impacts (A1-A3) dominate the life cycle.**
   In comparison to construction (A4-A5) impacts, production impacts make up the majority of the A1-A5 phase. MT1's substitution of structural concrete and steel with mass timber leads to substantial reductions across all impact categories.
2. **Use phase impacts are driven primarily by replacement.**
   Material replacement has larger contributions to the use phase than maintenance or repair, with high-impact materials like glass and steel shaping overall results.
3. **Use phase differences between the buildings arise entirely from enclosure materials**.

The building interiors are identical, making the enclosure the only source of potential reductions between the MTP and concrete designs. MT1's reduced glass use is a major driver of its lower use phase impacts.

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Chapter 2: Environmental Impacts of the End-of-Life Phase

Introduction

Compared with the growing body of cradle-to-gate and cradle-to-site research, far fewer studies examine the end-of-life (EOL) phase for MTP. Most MTP buildings are not yet old enough to reach EOL, and assumptions about EOL processes introduce substantial uncertainty. In a global review, 57 percent of WBLCA studies excluded EOL entirely (Duan et al., 2022). Only two North American cradle-to-grave studies were identified in the review, both analyzing the same 12-story building in Portland. Liang et al. (2021) reported a 12 percent reduction in embodied carbon when MTP was recycled at EOL, while Chen et al. (2020) reported a 20 percent reduction under a standard cradle-to-grave scope and a 70 percent reduction when system expansion was used to account for reuse, recycling, and recovery benefits.

These divergent findings highlight the sensitivity of cradle-to-grave results to underlying EOL assumptions. Liang et al. (2021) assumed a 70 percent recycling rate but excluded deconstruction and waste processing impacts due to limited data. Chen et al. (2020) included these processes but did not transparently document their LCI development. Together, these studies underscore the need for comprehensive, well-documented primary data for each EOL module.

Similar data gaps exist for other construction and demolition (C&D) materials. A recent review of U.S. EOL LCI data found that asphalt pavement was the only construction material with well-documented EOL data; most other materials lack comprehensive datasets (EPA, 2021). Given the novelty of MTP in the built environment, data for MTP-specific EOL processes are even less developed, necessitating deeper analysis.

The EOL phase consists of four modules: deconstruction or demolition (C1), transportation to waste processing facilities (C2), waste processing (C3), and disposal (C4). The following subsections examine each module in turn, addressing both broader C&D waste streams and MTP waste specifically.

Deconstruction/Demolition

Deconstruction or demolition—the first EOL module (C1)—involves dismantling a building using equipment such as excavators, loaders, skid-steers, and cranes (Athena, 1997). Demolition typically prioritizes speed and cost, whereas deconstruction aims to preserve material quality for reuse or recycling. Because MTP construction is relatively new (Riddle, 2023), few LCA studies have evaluated deconstruction practices for mass timber buildings. A review of 100 case study buildings found that only half reported demolition energy, with many studies omitting it on the grounds that it is negligible relative to earlier life cycle phases (Schenk & Amiri, 2022). This rationale contributes to the limited availability of demolition energy data and the absence of established methodologies.

Recent efforts have begun to address these gaps. Di Ruocco et al. (2023) developed a quantitative model estimating fuel consumption for removing connections, lowering materials, and stockpiling them. While this model fills an important need, no published studies have validated or replicated its findings. For demolition, the Athena Impact Estimator provides data for wood-framed structures, but comparable data for MTP buildings remain limited (Athena, 1997). As a result, no comprehensive comparative analyses of deconstruction and demolition have been conducted specifically for mass timber buildings.

Studies of other building types offer methodological insights that may be applicable to MTP. Quéheille et al. (2020) compared demolition and deconstruction for five concrete buildings and found that deconstruction resulted in lower embodied carbon, despite requiring more complex dismantling and a wider range of equipment. Their approach—multiplying daily machine energy consumption by equipment use duration—provides a potential framework for estimating impacts in MTP buildings. However, reproducibility is constrained by the lack of transparent reporting on equipment operation times.

Waste Transportation

After deconstruction, C&D waste is transported to processing facilities in the second EOL module (C2). Although C&D waste in the U.S. is nearly double the mass of municipal solid waste (MSW) (EPA, 2020), MSW transportation has been studied far more extensively. Challenges in characterizing C&D waste—such as material heterogeneity, bulkiness, and the absence of federal tracking requirements—contribute to this gap (Townsend et al., 2019).

MSW studies report transportation distances ranging from 23 kilometers (km) (EPA, 2023a) to 480–800 km for long-haul landfill transport (Thorneloe et al., 2005). Tracking studies in Seattle found an average distance of 113.95 km, generating 22.22 kg of carbon dioxide equivalent (CO_2e) per metric ton of MSW (Offenhuber et al., 2012). Comparable analyses for C&D waste are limited. Robinson & Kapo (2004) modeled relationships between recyclers, producers, and transportation networks in the Mid-Atlantic region, but did not report average distances, and the age of the data makes it unclear whether the facilities examined are still operating.

Existing EOL data sources typically model C&D waste transportation using a default distance of 32 km (EPA, 2024a). While this assumption simplifies modeling, it may not reflect regional variations in facility availability, routing, or waste management practices.

Waste Processing

Waste processing—the third EOL module (C3)—occurs at material recovery facilities (MRFs) and transfer stations, where C&D waste is unloaded, sorted, compacted, or baled (Eisted et al., 2009). Equipment such as forklifts, loaders, conveyors, and compactors is used throughout these operations. The purpose of waste processing is to separate recoverable materials from landfill-bound waste and to complete any cleaning, resizing, or preparation needed before materials can be recycled or incinerated.

Several studies assume that MTP waste can be reused, recycled into panel products, or incinerated with energy recovery (Santos et al., 2021; Salazar & Meil, 2009; Darby et al., 2012; Chen, 2019; Bjarvin et al., 2025). MTP's modularity and alignment with "Design for Disassembly" principles further support its potential for reuse (Cabrero et al., 2025). In reuse scenarios, MTP is typically processed at reuse facilities, where sanding, staining, or resizing may occur to prepare components for reinstallation.

However, existing U.S. LCI data for C&D waste processing is outdated, originating from studies conducted more than 20 years ago (Cochran, 2006; Levis, 2008). Other approaches—such as using product-specific EPDs (Silvestre et al., 2014) or generic LCA databases like DATASMART and ecoinvent (LTS, 2023; Wernet et al., 2014)—often omit waste processing emissions entirely, limiting their applicability for recycled materials. As a result, recent LCA studies on MTP have relied on proxy processes for reuse and recycling (e.g., trimming plywood, planing lumber) due to the absence of MTP-specific data (Bjarvin et al., 2025).

Waste Disposal

Waste disposal—the fourth and final EOL module (C4)—includes disposal processes that occur at incineration facilities or landfills. LCI data for C&D disposal is relatively well developed in databases such as DATASMART and ecoinvent (LTS, 2023; Wernet et al., 2024). Disposal impacts can also be modeled using the EPA Waste Reduction Model (WARM), which provides global warming potential (GWP_{FOSSIL}) emission factors for a range of C&D materials in different EOL scenarios and accounts for variations in moisture content and management practices (EPA, 2023a-b). However, WARM is limited in scope: it does not include other impact categories and does not provide detailed LCI data.

No disposal data specific to MTP currently exists, and it remains unclear whether MTP—with its resin content—is accepted at all incineration facilities. This uncertainty complicates modeling for incineration scenarios and highlights the need for facility-specific guidance. Resin content also influences landfill emissions, particularly for biogenic carbon accounting and potential methane generation, yet no standardized approach exists for modeling these effects for MTP.

Research Objectives

Many MTP building LCAs exclude EOL (C1–C4) impacts due to limited data and uncertainties in modeling long-term performance and waste management. Addressing the data gaps identified in existing C&D EOL LCA research is essential for improving understanding of MTP's comparative environmental performance relative to other building materials. This study expands upon previous EOL analyses by collecting primary data, validating assumptions through literature and surveys, and comparing environmental impacts across five EOL scenarios for MTP (reuse, reprocess, recycle, incinerate, and landfill) and three scenarios for other C&D materials (recycle, incinerate, and landfill).

The research objectives are:

1. **Collect primary data** on material and energy inputs for deconstructing a mass timber mock-up structure, and apply this model to the MT1 building.
2. **Apply excavator fuel consumption models** to estimate demolition impacts for the MT1 and Concrete Baseline buildings.
3. **Use logistics modeling** to estimate transportation distances and associated impacts for MTP waste, and validate the model through surveys. Replicate the model for all C&D waste for the city where the case study buildings are located.
4. **Collect primary data** on material and energy inputs for MTP waste processing in reuse and reprocess scenarios.

5. **Refine waste processing assumptions** for recycling, incineration, and landfill scenarios of all C&D materials using secondary data.
6. **Determine average resin contents** of U.S.-produced CLT and GLT and model disposal impacts using proportional resin-to-wood ratios; validate incineration feasibility through legislative review.
7. **Model waste disposal impacts** of all C&D materials using existing LCI data.
8. **Apply EOL phase impacts** of each material to an MTP case study building and an alternative concrete design to determine whole-building EOL impacts.

The overarching goal of this chapter is to assess the environmental impacts associated with C&D waste treatment across different U.S. end-of-life scenarios, thereby improving understanding of the environmental tradeoffs and benefits associated with each material and pathway. This research provides foundational data that can strengthen future LCA studies incorporating the EOL phase for MTP products or buildings, as well as other C&D materials.

Methods

Goal & Scope

This LCA was conducted in accordance with ISO 14040, ISO 14044, and ISO 21930 (ISO, 2006a-b; ISO, 2017). All impacts were modeled in SimaPro v9.5 using the TRACI characterization method (Bare, 2012). The functional unit for this analysis is one building.

The study includes all modules of the end-of-life (EOL) phase: C1 (building deconstruction), C2 (waste transportation), C3 (waste processing), and C4 (waste disposal). Figure 11 illustrates the system boundary and shows how processes and materials are included in each life cycle phase.

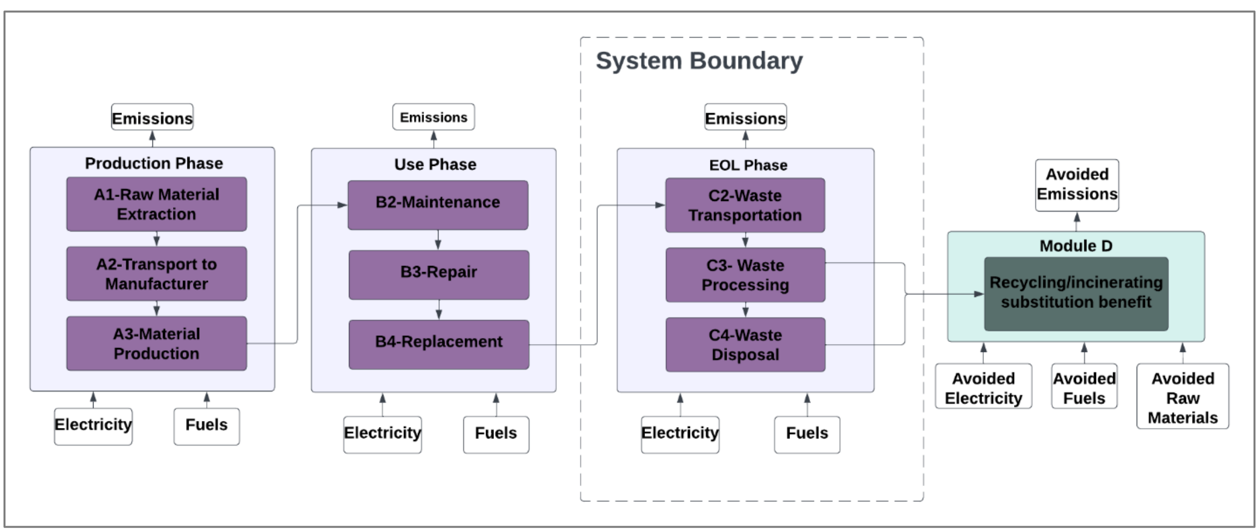


Figure 11. System boundary diagram for the end-of-life (EOL) phase of construction and demolition waste. Modules outside the dashed line are not included in the system boundary.

This study evaluates the five mandatory impact categories specified in ISO 21930: fossil global warming potential (GWP_{FOSSIL}), acidification potential (AP), eutrophication potential (EP), ozone depletion potential (ODP), and smog formation potential (SFP) (ISO, 2017).

C1–Demolition/Deconstruction

The first module of the EOL phase is dismantling a building through demolition or deconstruction (C1), with deconstruction involving more selective removal to preserve materials for reuse or recycling. This analysis includes three separate parts:

- **Mockup deconstruction**– collecting primary life cycle inventory (LCI) data during the deconstruction of a mass timber "mockup" structure.
- **MT1 deconstruction**– adapting the mockup LCI data to model the deconstruction impacts of the MT1 case study building.
- **Demolition**– modeling demolition impacts of MT1 and the Concrete Baseline by applying excavator fuel consumption models to each building.

Each part of this analysis is detailed in the following subsections.

Mockup Deconstruction

To address the data gap surrounding C1-Deconstruction impacts, a mass timber "mockup" structure provided an ideal case study (Figure 12). Constructed in 2017 as a large-scale replica of Google's MT1 building, the mockup was located at 1190 Borregas Avenue in Sunnyvale, California. The actual MT1 building, located nearby at 1265 Borregas Avenue, was completed in 2021. Mockups are commonly built prior to full construction to test structural systems, detailing, and constructability.

The Borregas mockup is a two-story, two-bay structure containing 102.23 m^3 of CLT and 47.71 m^3 of GLT. In comparison, the five-story MT1 building contains 4,292.67 m^3 of CLT and 1,857.07 m^3 of GLT across its structure and enclosure. The mockup incorporated several key MT1 structural elements, including columns, beams, and a composite cantilever deck composed of CLT embedded with HBV® metal mesh and topped with a concrete slab.

Figure 12. Mockup at 1190 Borregas, Sunnyvale, California. Photo by Christina Bjarvin.

Challenges and Obstacles

Deconstruction followed the material disassembly sequence outlined in the structural and erection drawings. XL Construction documented several challenges encountered during the process (Beck, 2024):

- **Unaccounted screws:** Additional screws used during construction were not reflected in the drawings, slowing removal.
- **Broken long screws:** When ~24-inch screws fractured during extraction, hydraulic jacks were required to separate CLT panels from GLT beams, damaging the top layer of the GLT and rendering those beams non-reusable.
- **Composite deck fasteners:** In the cantilever deck, HBV® mesh was attached using epoxy and screws. Screw heads were damaged during concrete removal, complicating extraction.
- **Column baseplates:** Baseplates were smaller than the GLT columns, leaving insufficient clearance for pneumatic tools to remove anchor bolt nuts efficiently.
- **Material damage during removal:** Several third-level glulam beams and blocking pieces were damaged or delaminated during CLT removal. These materials were sent to a waste facility; however, they represented only 1.3% of the total MTP volume, meaning 98.7% of the structure's MTP was successfully recovered for reuse.

Equipment

XL Construction and Elevated Construction Services performed the deconstruction with the goal of diverting MTP from landfill toward reuse or recycling. The team documented:

- equipment types used for each deconstruction activity
- onsite energy consumption
- quantities of MTP recovered for reuse

This documentation enabled a comprehensive life cycle inventory (LCI) for the C1 module.

Table 6 lists the equipment used. Power tools and scissor lifts were powered through a spider box, for which the deconstruction team provided submeter readings. The structure was fully deconstructed in eight days, with additional pre- and post-deconstruction activities occurring over the following three weeks.

Table 6. Types of equipment used during the deconstruction of the 1190 Borregas mockup.

Equipment	Use
Saw cutters	Cutting up concrete deck
Jackhammers	Breaking up concrete deck
Impact drivers	Screw and connector removal
Hydraulic jacks	Separating CLT panels from GLT beams to expose broken screws
Scissor lifts	Transporting crew and equipment to second floor and roof
Boom lift	Transporting concrete to ground

Crane (Model RTC-8090)	Transporting steel and MTP to ground
Forklifts (5k and 26k)	Transporting materials on ground to flatbed truck; loading and unloading materials on truck
Flatbed truck	Delivering equipment to jobsite

The full LCI for deconstruction is shown in Table 7. Electricity consumption for power tools and scissor lifts was directly measured. Fuel consumption for the boom lift, crane, forklifts, and flatbed truck was estimated based on equipment use duration and the mass of materials handled, using the methods described in the subsections below.

Table 7. Life cycle inventory of deconstruction activities for the mockup structure.

Input Description	Quantity	Unit	LCI Database
Diesel, for crane	255.46	L	US-EI 2.2
Diesel, for boom lift	37.99	L	US-EI 2.2
Diesel, for forklifts	17.58	L	US-EI 2.2
Electricity, for power tools and scissor lifts	47.00	kWh	ecoinvent
Transport, equipment delivery	6,468.77	tkm	US-EI 2.2

Boom Lift and Crane

The boom lift transported broken concrete from the composite deck to the ground. Diesel consumption was estimated using the fuel model in Equation 1 (Chapter 1), applied to the mass of the concrete. The same model was used to estimate crane fuel consumption for transporting steel and MTP from the deck and roof to the ground. Of the crane's total 255.46 L of diesel use, 62 percent was attributable to MTP handling.

Forklifts

Both a 5k forklift (2.26-ton capacity) and a 26k forklift (11.79-ton capacity) were used to move materials across the site, stack them, and load them onto a flatbed truck. Diesel consumption was calculated using Equation 2.

Equation 2. Diesel fuel consumption for forklifts

$$F = \frac{t * d * c}{v}$$

Where:

- F = diesel consumption (L)
- t = number of round trips
- d = loading distance (km)
- c = hourly fuel consumption (L/hr)
- v = forklift speed (km/hr)

Values for each variable are shown in Table 8. Total forklift diesel consumption was 17.58 L.

Table 8. Values for variables used to determine diesel fuel consumption by 5k and 26k forklifts during the deconstruction of the Borregas mockup.

Variable	Unit	5k Forklift Value	26 Forklift Value	Source
Number of round-trips (*t*)		26	39	
Loading distance (*d*)	km	0.24	0.24	Google Maps
Hourly fuel consumption (*c*)	L/hour	3.2	12.5	"What is the fuel consumption per hour for a diesel forklift?," n.d.
Forklift speed (*v*)	km/hour	8.05	8.05	"1910.178 - Powered industrial trucks," n.d.
Total diesel consumption (*F*)	L	2.54	15.04	

The number of forklift trips was determined by:

1. Calculating the mass of each piece of MTP, steel, and concrete.
2. Assigning each piece to the appropriate forklift based on load capacity.
3. Summing the daily cumulative mass handled by each forklift.
4. Dividing cumulative mass by forklift load capacity.

This resulted in 26 trips for the 5k forklift and 39 trips for the 26k forklift.

Equipment Delivery

All equipment listed in Table 6 was rented from Arrow Rental & Supply (Fremont, CA), except the crane, which was rented from Bigge Crane and Rigging Co. (San Leandro, CA). Equipment masses were obtained from specification sheets and multiplied by round-trip distances to the jobsite to calculate transport ton-kilometers (6,458.77 tkm). All deliveries were assumed to use flatbed trucks.

Sensitivity Analysis

To evaluate the robustness of the fuel-based modeling approach, a sensitivity analysis compared the original model to alternative methods used in published LCA studies (Di Ruocco et al., 2023; Quéheille et al., 2020; Cole, 1998). Instead of fuel consumption, the sensitivity model used equipment operating hours as inputs, following Hemmati et al. (2024). Emission factors from ecoinvent's machine operation processes were applied to estimate environmental impacts (Wernet et al., 2014).

Electricity consumption for scissor lifts and power tools remained unchanged because primary data was available. The machine-operation approach was applied only to the crane, forklifts, and boom lift. Table 9 presents the LCI used in the sensitivity analysis.

Table 9. Life cycle inventory of deconstruction activities for the mockup structure in a sensitivity analysis.

Description	Value	Unit	Process Name	LCI Database

Duration of machine use, for crane	24.42	hr	Machine operation, diesel, >= 74.57 kW, high load factor {GLO}| market	ecoinvent
Duration of machine use, for boom lift	2.80	hr	Machine operation, diesel, >= 18.64 kW and < 74.57 kW, high load factor {GLO}| market	ecoinvent
Duration of machine use, for forklifts	1.99	hr	Machine operation, diesel, >= 18.64 kW and < 74.57 kW, high load factor {GLO}| market	ecoinvent
Electricity, for power tools and scissor lifts	47	kWh	Electricity, low voltage {WECC, US only}| market	ecoinvent
Transport, equipment to deconstruction site	6,468.77	tkm	Transport, single unit truck, diesel powered NREL/US U	US-EI 2.2

Crane operating hours were estimated using an average uptime of 43.6% for telescopic cranes (Lawani et al., 2020), applied to a seven-day rental period (8 hours/day), yielding 24.42 hours of use. Forklift operating hours (1.99 hours) were carried over from the original model.

Boom lift operating hours were estimated in two phases:

1. **Vertical extension** between ground and deck
   - Extension speed: 548.64 meters/hour (m/hr)
   - Round-trip vertical distance: 8.53 m
   - Total trips: 75
   - Total time: 1.17 hr
2. **Manual loading of concrete (Equation 3)**
   - Round-trip loading distance: 13.41 m
   - Walking speed: 5,040 m/hr
   - Time per trip: 10 seconds (s)
   - Number of trips: 612 (based on 27.5 kilograms (kg) recommended shoveling load)
   - Total time: 1.63 hr

Combined, the boom lift operated for 2.8 hours.

Equation 3. Loading time calculation

$$T = \frac{d}{v}$$

Where:

- T = loading time (hr)
- d = loading distance (m)
- v = loading velocity (m/hr)

MT1 Deconstruction

The findings of the mockup deconstruction analysis can be scaled to the full-building level for MT1. Although the mockup replicated MT1's structural assemblies, it did not include enclosure or interior materials and represented only a small fraction of the full building. As a result, the mockup-based methods must be adapted to estimate whole-building C1 impacts for MT1.

The mockup was deconstructed entirely by hand: jackhammers and saw cutters were used to break apart the composite cantilever deck, concrete debris was loaded manually onto lifts for ground transport, and impact drivers were used to remove screws from the MTP. These methods enabled a high MTP recovery rate (>98%), but the deconstruction team emphasized that such labor-intensive practices would not be feasible at full building scale. Nevertheless, the mockup results provide a basis for developing a hand-deconstruction model for MT1 that can be compared directly to the conventional demolition model. In this scenario, deconstruction is used specifically to expose and recover MTP in the composite deck assembly for reuse or recycling.

At full scale, deconstruction of MT1 would begin with removal of enclosure and interior components using hydraulic excavators, similar to the demolition scenario. Excavators would also be used to remove structural materials other than the composite cantilever deck.

Once non-structural and non-deck materials are removed, the next step would be to dismantle the composite cantilever deck to expose the CLT floor panels. The deck consists of CLT panels with HBV© mesh epoxied into the wood, topped with reinforced concrete. This composite bond is significantly stronger than a typical CLT + concrete topping assembly, making removal more challenging. Based on input from XL Construction and a Sunnyvale-based deconstruction expert, a gasoline-powered concrete saw—not jackhammers—would be used at full scale to cut the 3.5-inch concrete topping slab into 5 ft × 5 ft panels. A skid steer would then haul the cut panels to a crane for lowering to the ground.

After the composite deck is removed, the structural MTP (CLT floor panels and GLT beams) would be recovered using the same equipment used in the mockup deconstruction. Screws would be removed with impact drivers, and hydraulic jacks would be used to separate CLT panels from GLT beams when screws are damaged. Because electrical tools contributed less than 1% of total C1 impacts in Chapter 1, these processes are excluded from the MT1 LCI. All recovered MTP would be lowered to the ground using a telescopic crane.

Once the above-grade structure is fully deconstructed, the below-grade structure (foundation) would be demolished using hydraulic excavators (Athena, 1997). Forklifts would then sort and stockpile all materials on the ground and load them onto trucks for waste transportation. The mockup deconstruction used one 5k and one 26k forklift; however, for MT1, a conservative assumption is applied: two 5k forklifts are used, since a 26k forklift would require less fuel to move the same mass of materials.

To estimate the total deconstruction energy consumption for MT1, the following modeling approach is applied:

- **Concrete saw**: Develop a gasoline consumption model for cutting the composite deck.
- **Skid steers:** Apply Athena's diesel consumption factor for concrete slab removal (2.92E-03 L/kg) to the mass of concrete in the composite deck (Athena, 1997).
- **Excavators**: Apply the Average demolition model's diesel consumption factor (4.72E-03 L/kg) to all materials except the concrete in the deck.
- **Crane:** Apply the crane diesel consumption model (Equation 1, Chapter 1) using the mass of structural MTP and MT1's building height.
- **Forklifts**: Apply the forklift fuel consumption model (Equation 2, Chapter 1) to the total mass of materials.
- **Equipment transport**: Model round-trip transport of each piece of equipment from the rental facility to MT1 using a ton-distance approach.

A gasoline consumption model for the concrete saw was developed using assumptions about cutting speed, hourly fuel consumption, and the physical characteristics of the topping slab. Equation 4 expresses gasoline consumption per kilogram of concrete:

Equation 4. Gasoline consumption of a concrete saw cutter.

$$G = \frac{2 * F}{v * s * t * p},$$

Where:

- *G* = gasoline consumption (L/kg),
- *F* = hourly fuel consumption (L/hr),
- *v* = cutting speed (m/h)
- *s* = cut spacing (m),
- *t* = slab thickness (m),
- *p* = concrete density (kg/m^3)
- *2* = factor to account for two cuts per 5'x5' panel

Using an average fuel consumption of 2 L/hour, a cutting speed of 70 m/hour (after reductions for HBV© mesh and CLT protection), a cut spacing of 1.52 m, a slab thickness of 0.089 m, and a concrete density of 2,400 kg/m^3 yields a consumption of 1.46E-5 L/kg of concrete.

This value is applied to the mass of concrete in the composite deck. The resulting life cycle inventory for deconstructing MT1—including concrete saw gasoline, skid steer diesel, excavator diesel, crane diesel, forklift diesel, and equipment transport—is presented in Table 10.

Table 10. Deconstruction life cycle inventory inputs for the MT1 building to salvage MTP in the structure for reuse.

Input	Quantity	Unit	Description
Gasoline, concrete saw	13.40	L	Breaking up concrete deck assembly
Diesel, skid steer	2,683.44	L	Breaking up concrete deck assembly
Diesel, excavators	47,862.32	L	Removal of remaining materials
Diesel, crane	6,459.12	L	Transportation of structure MTP to ground
Diesel, forklifts	575.23	L	Stockpiling and loading all materials on ground

Truck transportation, equipment	9,973.53	tkm	Round-trip transportation of equipment

The deconstruction model is not applied to the Concrete Baseline, as the primary purpose of deconstructing MT1 is to increase the amount of structural material recovered for reuse. MTP's modularity and compatibility with "Design for Disassembly" principles support its potential for reuse (Cabrero et al., 2025). Conversely, reuse of reinforced concrete is inhibited by several factors, including the lack of regrading techniques to assess structural integrity, potential cracking during dismantling, and corrosion of embedded rebar over time (Devènes et al., 2024; Atkins & Lambert, 2022). Additionally, the layout of rebar within cast-in-place concrete is highly specific to individual building load requirements (Rahimi & Maghrebi, 2023), which limits direct reuse of reinforced concrete as a whole unit. The mechanical bond between rebar and concrete also inhibits reuse of either component individually, since removing either material would inevitably damage both.

Building Demolition

The alternate dismantling model applied to both buildings is demolition using conventional mechanical methods. Excavators or skid-steer loaders separate assemblies, and cranes lower materials to the ground. Crane fuel use is modeled using Athena's crane diesel consumption equation (Equation 1, Chapter 1), which relates fuel consumption to the mass of materials handled and the average building height.

To capture variability in published demolition data, three independent excavator energy models were evaluated:

1. **Athena (1997)** – kg-basis and m^2-basis models for wood-framed and concrete-framed structures
2. **Schenk & Amiri (2022)** – literature-based averages for wood and concrete structures
3. **DATASMART (LTS, 2023)** – material-specific diesel consumption for concrete, reinforced concrete, gypsum, and rebar

Each model was applied to the MT1 and Concrete Baseline buildings using either total mass, gross floor area, or material-specific masses, depending on the model's functional unit. Diesel consumption was normalized to liters (L) per kilogram of building material to enable comparison, and an "Average" model was created by averaging the four model results (Table 11).

Table 11. Diesel consumption of excavators in liters per kilogram of material during building demolition in two case study buildings. The diesel consumption of five different demolition energy models is presented for comparison.

Model	L diesel/kg (MT1)	L diesel/kg (Concrete Baseline)
Athena (kg-basis)	1.09E-02	2.78E-03
Athena (m^2-basis)	3.82E-03	2.54E-03
Literature Review	1.84E-03	3.16E-03
DATASMART	2.32E-03	2.44E-03
Average	4.72E-03	2.73E-03

In addition to excavator fuel, two other demolition processes were included to fully represent C1 impacts:

- **Forklift use:** Forklifts are used onsite to sort, stockpile, and load materials onto refuse trucks. Forklift diesel consumption was calculated using Equation 2, developed in Chapter 1 for the 1190 Borregas mockup deconstruction. This ensures consistency between the mockup-based modeling framework and the whole-building demolition model.

- **Equipment transportation:** Transportation of demolition equipment to and from the site was modeled using a ton-distance approach, in which the total mass of equipment is multiplied by the round-trip distance between the equipment supplier and the demolition site. This captures the upstream fuel use associated with mobilizing excavators, forklifts, and other machinery.

For both buildings, it was assumed that two forklifts and four excavators were used during demolition, reflecting the equipment mix observed during MT1's construction.

Table 12 summarizes the life cycle inventory inputs for all demolition processes, including excavator diesel (for each model), forklift diesel, and equipment transport.

Table 12. Demolition life cycle inventory inputs for two case study buildings, with excavator inputs varying based on the excavator energy model used. The forklift use and equipment transportation inputs are equivalent regardless of the excavator model used, and are added to each excavator model's inputs to yield the total C1-Demolition impacts for each excavator model.

Input	Quantity (MT1)	Quantity (Concrete Baseline)	Unit	Description
Diesel, excavator, Athena (kg-basis) model	147,527.35	49,458.55	L	Separation and removal of materials from building
Diesel, excavator, Athena (m^2-basis) model	51,665.79	45,149.19	L	Separation and removal of materials from building
Diesel, excavator, Literature Review model	24,941.55	56,118.48	L	Separation and removal of materials from building
Diesel, excavator, DATASMART model	17,865.75	32,085.24	L	Separation and removal of materials from building
Diesel, excavator, Average model	60,500.11	45,702.86	L	Separation and removal of materials from building
Diesel, forklifts	575.23	755.14	L	Stockpiling, loading materials on trucks
Truck transportation, equipment	4,657.22	4,657.22	tkm	Round-trip transportation of equipment

C2–Waste Transportation

The next module in the EOL phase is waste transportation (C2) of materials after building deconstruction or demolition to facilities that will process, sort, and ship out waste to the next destination. The transportation network for C&D waste involves multiple steps that vary by EOL scenario. C&D waste typically enters the waste stream through three pathways:

1. Construction contractors delivering self-hauled waste to certified C&D processing facilities,
2. Private third-party haulers transporting waste to these facilities, and
3. Municipal waste collectors delivering C&D waste as part of city-contracted services ("Seattle's 2022 Solid Waste Plan," 2022).

C&D waste processing facilities include material recovery facilities (MRFs), transfer stations, recyclers, and reuse centers.

MRFs sort and temporarily store incoming waste to maximize diversion from landfills. Sorting may occur through automated lines, mechanical equipment, or manual labor ("City Terrace Recycling," n.d.). Recyclable materials are separated and sent to recyclers, while residual waste is sent to landfills. Some MRFs also divert materials to reuse centers.

Transfer stations weigh, unload, screen, sort, compact, and transport waste to disposal facilities such as landfills or mass-burn incinerators. Although the EPA distinguishes MRFs from transfer stations based on whether recyclable materials are separated (EPA, 2002), many facilities labeled as transfer stations now perform sorting activities. This suggests that the EPA's definitions may be outdated and that the terms are increasingly interchangeable in practice.

Recyclers further process wood waste by removing contaminants, shredding and screening the material, and producing chips of consistent size ("Wood," n.d.). These chips are sold for use in mulch, paper, cardboard, or biofuel. Transportation to the next user is considered outside the EOL system boundary and is therefore not included in this analysis.

Reuse centers accept self-hauled waste or diverted materials from MRFs and transfer stations. Some reuse centers conduct onsite processing—such as sanding, staining, resizing, or de-nailing—while others sell materials without additional processing. Under both the reuse and reprocess EOL scenarios, waste processing includes de-nailing and resizing panels. The reprocess scenario additionally includes planing or sanding to prepare MTP for lower-grade aesthetic applications. The key distinction is that reused MTP retains its original form and intended structural or non-structural use, whereas reprocessed MTP is used in lower-grade applications such as countertops or shelving (Robertson et al., 2012). Both reused and reprocessed MTP are sold through reuse centers.

Figure 13 illustrates the flow of waste through the transportation network across the five EOL scenarios.

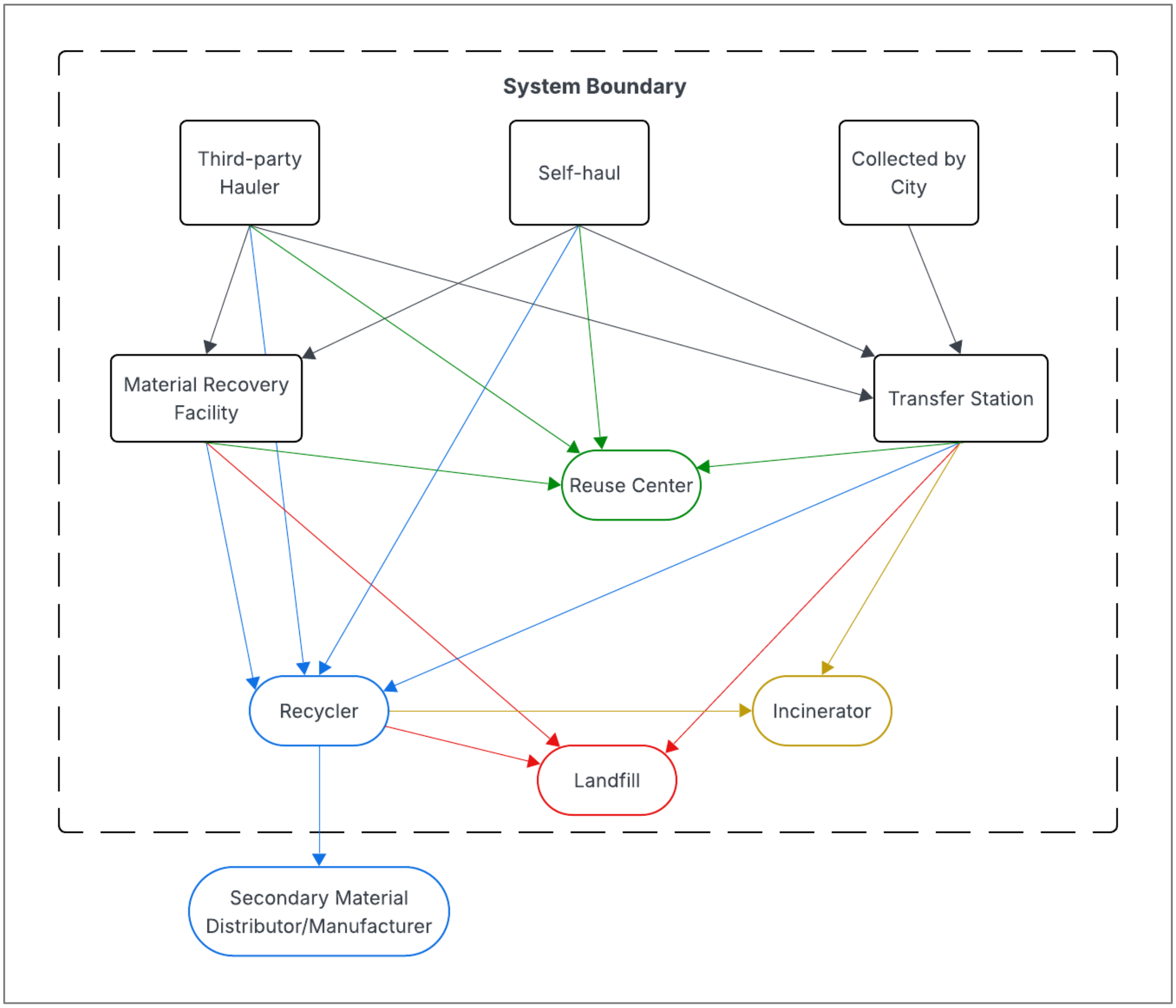


Figure 13. Visualization of the construction and demolition waste transportation network in five end-of-life scenarios: reuse and reprocess (green), recycle (blue), incinerate (yellow), and landfill (red).

This analysis includes three separate parts:

- **MTP ArcGIS model**– geospatial modeling of nationwide waste transportation for MTP.
- **MTP transport survey**– validating the geospatial transportation model for MTP through surveys in five major cities.
- **Sunnyvale transport survey**– expanding the MTP transport survey to other C&D materials for the city of Sunnyvale, California (where the case study buildings are located).

Each part of this analysis is detailed in the following subsections.

MTP ArcGIS Model

To address the lack of research on transportation impacts for C&D wood waste such as MTP, logistics modeling was conducted using the Closest Facility tool in ArcGIS v3.2. This tool identifies the shortest travel route between specified starting points and the closest destinations. In this analysis, starting points were waste-generating areas (cities), intermediate points were waste processing facilities (MRFs, transfer stations, and recyclers), and destinations were disposal facilities (incinerators and landfills) (Figure 14). For the reuse, reprocess, and recycle scenarios, reuse centers and recycling centers served as the final destinations, with no intermediate processing facilities modeled.

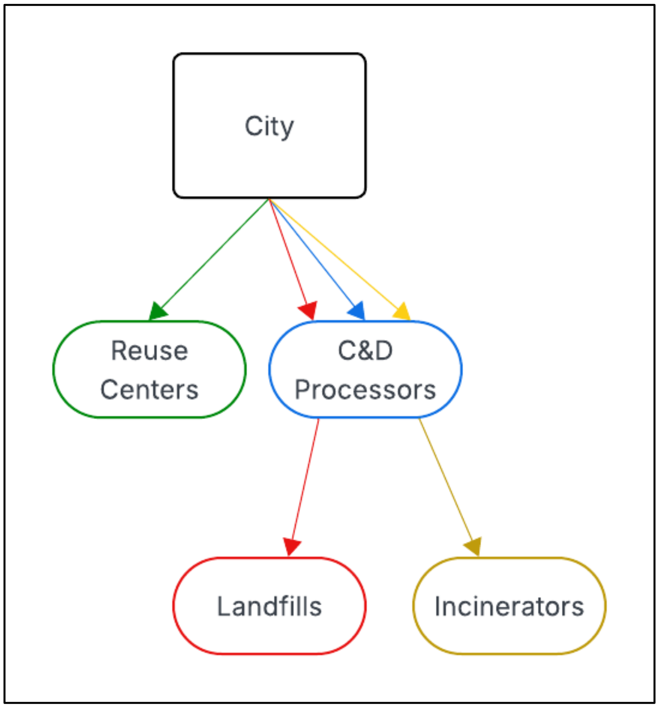


Figure 14. ArcGIS modeling pathway for transportation between cities and waste processing/disposal destinations for five end-of-life scenarios: reuse and reprocess (green), recycle (blue), incinerate (yellow), and landfill (red).

Three datasets were compiled for the model: major waste-generating areas, waste processing facilities, and disposal facilities (

Table 13). Cities with populations greater than 10,000 (2020 Census) were selected as proxies for major waste-generating areas, representing 60% of the U.S. population.

Table 13. ArcGIS layer sources for cities and waste processors.

Layer	Source	Number of Facilities	Layer Modifications
Cities	esri_dm, 2024	3,886	-Cities with populations <10,000 removed
Reuse Centers	“Reuse Ecosystem Map,” n.d.	524	-Canadian facilities removed - “Reuse,” “Deconstruction/Salvage,” “Hauling/Warehouse,” and “Remanufacturing/Recycling” facilities used only
Recycling Infrastructure	EPA_GEO, 2024	1,864	-Landfills removed -Non-C&D facilities removed
Incinerators	EPA & OLEM, 2021	72	- “HW,” “CISW,” and “HMIW” incinerators removed
Landfills	Oak Ridge National Laboratory, 2017	2,308	-Closed landfills removed

The recycling infrastructure layer includes recyclers and waste processing facilities (MRFs and transfer stations). All facilities in this layer were treated as intermediate destinations for waste that would eventually be incinerated or landfilled, since non-recyclable materials are sent to disposal after sorting. Transportation between processing facilities and recyclers was not modeled due to data limitations; instead, it was assumed to be negligible, consistent with MSW analyses that assume transfer stations are located near waste-generation areas or disposal sites (Thornloe et al., 2005).

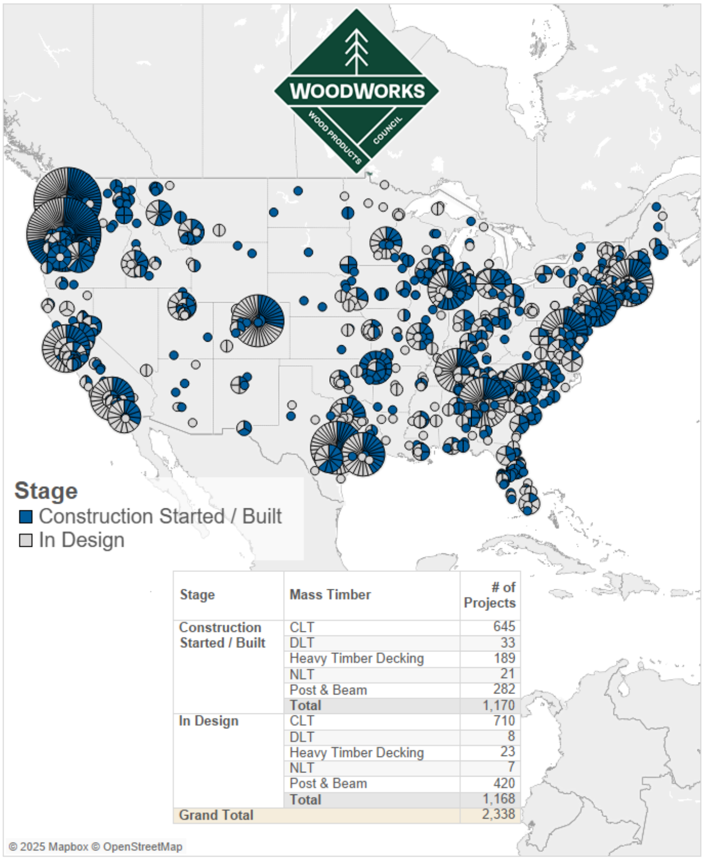
Trucks were selected as the transportation mode. The 2017 U.S. Commodity Flow Survey shows that trucks are the dominant mode for waste transport by both tonnage and value (United States Census Bureau, 2017). International studies similarly report trucks as the preferred mode for C&D waste due to ease of loading and flexible scheduling (Rosado et al., 2022). The route network used in ArcGIS was developed by the U.S. Department of Transportation for freight truck transport (Bureau of Transportation Statistics & Federal Highway Administration, 2017).

The Closest Facility tool generated thousands of routes between cities and the closest facilities. For each facility or city sending out waste, three facilities were selected to receive waste to avoid underestimating distances in cases where the nearest facility is not used. Route distances were calculated using the “Calculate Geometry” function, and mean distances were obtained using “Visualize Statistics.” A 500-mile cutoff was applied to avoid unrealistic long-distance transport. Environmental impacts of transporting 1 kg of MTP waste were modeled using the DATASMART truck transport process in SimaPro (LTS, 2023). A sensitivity analysis was conducted by reducing the number of receiving facilities from three to one, generating a single route to the closest facility for each scenario.

MTP Transport Survey

To validate the ArcGIS model, surveys were conducted in the largest city of the five U.S. states with the highest number of MTP projects: Seattle, WA; Portland, OR; Los Angeles, CA; Boston, MA; and Houston, TX (Figure 15). Both “in design” and “construction started/built” projects were considered.

Mass Timber Projects In Design and Constructed in the US (December 2024)



Stage	Mass Timber	# of Projects
Construction Started / Built	CLT	645
	DLT	33
	Heavy Timber Decking	189
	NLT	21
	Post & Beam	282
	Total	1,170
In Design	CLT	710
	DLT	8
	Heavy Timber Decking	23
	NLT	7
	Post & Beam	420
	Total	1,168
Grand Total		2,338

State	Stage	
AK	In Design	8
AL	Construction Started / Built	14
	In Design	10
AR	Construction Started / Built	26
	In Design	9
AZ	Construction Started / Built	3
	In Design	5
CA	Construction Started / Built	140
	In Design	162
CO	Construction Started / Built	37
	In Design	50
CT	Construction Started / Built	21
	In Design	10
DC	Construction Started / Built	12
	In Design	19
DE	Construction Started / Built	2
	In Design	1
FL	Construction Started / Built	39
	In Design	33
GA	Construction Started / Built	26
	In Design	55
HI	Construction Started / Built	3
	In Design	5
IA	Construction Started / Built	11
	In Design	2
ID	Construction Started / Built	14
	In Design	13
IL	Construction Started / Built	25
	In Design	21
IN	Construction Started / Built	11
	In Design	7
KS	Construction Started / Built	3
	In Design	5
KY	Construction Started / Built	9
	In Design	12
LA	Construction Started / Built	4
	In Design	7
MA	Construction Started / Built	48
	In Design	65
MD	Construction Started / Built	11
	In Design	26
ME	Construction Started / Built	17
	In Design	10
MI	Construction Started / Built	19
	In Design	32
MN	Construction Started / Built	17
	In Design	22
MO	Construction Started / Built	12
	In Design	18

State	Stage	
MS	Construction Started / Built	2
	In Design	13
MT	Construction Started / Built	21
	In Design	11
NC	Construction Started / Built	50
	In Design	41
ND	Construction Started / Built	3
	In Design	1
NE	Construction Started / Built	7
	In Design	11
NH	Construction Started / Built	6
	In Design	11
NJ	Construction Started / Built	13
	In Design	15
NM	Construction Started / Built	3
	In Design	3
NV	Construction Started / Built	1
	In Design	4
NY	Construction Started / Built	39
	In Design	50
OH	Construction Started / Built	19
	In Design	22
OK	Construction Started / Built	5
	In Design	9
OR	Construction Started / Built	127
	In Design	43
PA	Construction Started / Built	16
	In Design	18
RI	Construction Started / Built	8
	In Design	1
SC	Construction Started / Built	26
	In Design	11
SD	Construction Started / Built	4
	In Design	1
TN	Construction Started / Built	17
	In Design	37
TX	Construction Started / Built	79
	In Design	101
UT	Construction Started / Built	16
	In Design	23
VA	Construction Started / Built	19
	In Design	33
VT	Construction Started / Built	6
	In Design	15
WA	Construction Started / Built	120
	In Design	73
WI	Construction Started / Built	34
	In Design	13
WV	Construction Started / Built	2
WY	Construction Started / Built	3
	In Design	1

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help@woodworks.org
woodworks.org/project-assistance

Figure 15. Total number of MTP projects (planned and built) by state ("Mass Timber Projects in Design & Constructed," 2025).

For each city, certified C&D processing facilities (transfer stations, MRFs, recyclers, and reuse centers) were identified through municipal public utilities websites. Transfer stations were contacted to determine their disposal destinations. Distances between city centers and transfer stations, and between transfer stations and disposal facilities, were measured using Google Maps; the longest available route was selected as a conservative estimate. Survey questions are provided in Appendix B.

MRFs were contacted to determine whether they accept CLT or GLT. Only facilities accepting MTP were included. Facilities were asked where they send processed wood waste, whether they recycle onsite, their diversion rates, and whether outgoing waste is shipped by truck or rail. Distances between cities and MRFs, and between MRFs and their next users, were mapped using Google Maps.

Recyclers were contacted to determine whether they accept MTP, their diversion rates, and their landfill destinations. If a facility declined to identify a landfill, the largest C&D landfill in the state (by annual throughput) was assumed.

Reuse centers were contacted to determine whether they accept MTP and whether size restrictions apply. Facilities were identified using the Building Product Ecosystems reuse map, consistent with the ArcGIS reuse layer ("Reuse Ecosystem Map," n.d.).

Survey results were used to calculate total transportation distances for each EOL scenario. Total tonne-kilometers (tkm) were calculated using Equation 5.

Equation 5. Total transportation distance in tonne-kilometers (tkm)

$$tkm = \sum_{i=1}^{n} D_i * W$$

Where:

- D_i = transportation between facilities (km)
- W = mass of waste transported (MT)

Because primary data on mass losses during transportation is unavailable, waste diversion reports were reviewed to assess whether inbound and outbound wood tonnages could be used to estimate losses. However, differences in waste categorization and multiple outbound destinations made it impossible to determine true mass loss. As a simplifying assumption, all mass loss was attributed to waste processing (C3), not transportation.

Sunnyvale Transport Survey

The transportation analysis conducted for MTP in five major U.S. cities in Chapter 1 was replicated for all C&D materials in Sunnyvale, CA. The city's C&D waste management website provided a list of approved waste processing facilities. Each facility was surveyed to determine:

- the types of waste accepted,
- the EOL treatment options available for each material (e.g., recycling, landfilling),
- the outbound transportation mode, and
- the distance to downstream facilities receiving outbound waste.

The Sunnyvale analysis showed that C&D wood waste has four available EOL pathways: reuse, recycling, incineration, and landfilling. Concrete, metals, and gypsum can be either recycled or landfilled. All other C&D materials used in MT1 are landfilled. All transportation was reported to occur by truck. The transportation distances were applied to each material in a ton-distance approach

C3–Waste Processing

The third module of the EOL phase is waste processing (C1), which typically occurs in material recovery facilities (MRFs), transfer stations, recyclers, or reuse facilities. Waste processing can also happen at deconstruction or demolition sites, but in this analysis, all processing is assumed to occur at downstream processing facilities.

The material and energy inputs for waste processing vary by EOL scenario and material type. In this analysis, five EOL scenarios are assumed for MTP (reuse, reprocess, recycle, incinerate, and landfill), and three scenarios are assumed for other C&D materials (recycle, incinerate, and landfill).

This analysis includes two separate parts:

- **Reuse center processing**– collecting primary LCI data from a reuse center for MTP in the reuse and reprocess EOL scenarios.
- **MRF processing**– applying secondary LCI data from a material recovery facility (MRF) for recycling, incineration, and landfill scenarios of all C&D materials.

Each part of this analysis is detailed in the following subsections.

Reuse Center Processing

The reuse and reprocess scenarios for MTP involve retaining MTP in its original form after deconstruction. In the reuse scenario, panels may be directly incorporated into new buildings as structural or non-structural elements. In the reprocess scenario, panels are converted into lower-grade products such as benches, butcher-block countertops, or garden beds. To estimate the environmental impacts of waste processing for both scenarios, data was collected from key participants in the construction circular economy.

Five reuse centers in the Seattle metropolitan area were identified using the Building Product Ecosystems reuse map: Reuse Innovation Center (RIC), Ballard Reuse, Urban Reclamations, Second Use, and Earthwise ("Reuse Ecosystem Map," n.d.). Each facility was interviewed to determine whether they accept GLT, what types of processing are required before resale, and whether processing occurs onsite or elsewhere. Facilities were also surveyed about the proportion of GLT used in building versus non-building applications and criteria for determining reuse suitability.

Of the five centers, three accept MTP: Second Use, Earthwise, and RIC. Second Use does not conduct waste processing onsite; donated materials are sold as-is, and purchasers cut beams to size themselves. Earthwise and RIC both process salvaged GLT onsite. All three facilities reported that salvage contractors rely on visual grading to determine whether GLT beams are suitable for reuse.

Second Use estimated that 60% of GLT beams sold are used in building products and 40% in non-building products, with benches being the most common. Structural reuse typically involves "overdesigning," where beams are used under lower stress demands than their original design loads. Structural integrity is either evaluated by a lumber engineer or addressed through conservative design.

Earthwise conducts de-nailing, milling, and—under the reprocess scenario—planing and sanding. Reprocessed GLT is marketed for butcher block countertops and shelving, while reused beams may be sold in their original dimensions. Earthwise estimated that 90–95% of GLT sold is used as non-structural framing material. Washington State building codes classify reused wood as #2 or higher for both residential and commercial applications, allowing structural reuse without requiring re-grading by an engineer.

RIC also processes GLT onsite and reported a wide range of next uses, including small structures, bridge supports, countertops, benches, garden beds, shelving, tables, and aesthetic beams. Cutting beams to customer-specified dimensions is the most common processing activity; reprocessed beams often require sanding and staining for aesthetic applications.

Because most donated MTP arrives directly from deconstruction sites, MRF or transfer-station processing is not included in the C3 impacts for reuse or reprocess scenarios. Instead, impacts were estimated using primary data collected from Earthwise, which operates the King County Salvage Lumber Warehouse (SLW), a three-year pilot project processing approximately 20 MT of salvaged wood per month ("Circular economy," n.d.). Figure 16 and Figure 17 show salvaged GLT before and after processing.

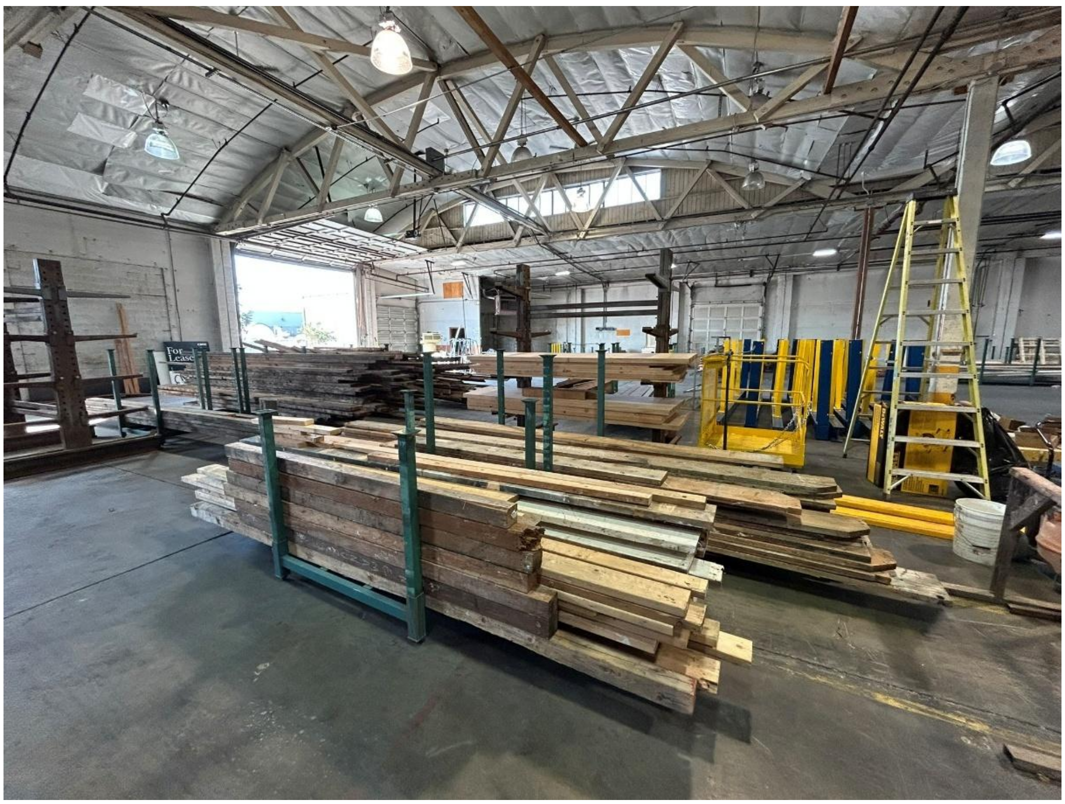

Figure 16. Salvaged lumber and glue-laminated timber (GLT) at the King County Salvage Lumber Warehouse.

Figure 17. Salvaged glue-laminated timber (GLT) after milling and planing.

A batch of salvaged GLT (286 board feet (BF)) was used as a case study. The beams were originally 20 feet long, cut in half, and planed for use as butcher-block countertops. BF were converted to cubic meters using a regional average of 417 BF/m^3, and to mass using a regional GLT density of 511 kg/m^3 (Bowers et al., 2019).

Earthwise reported 2 hours of portable sawmill use and 8 hours of planer use. The sawmill (Wood-Mizer LT40 Hydraulic, 26.5 HP gas engine) consumed 7.38 L/hr of gasoline, calculated using a brake-specific fuel consumption of 0.45 lb/hp-hr ("Consumption Calculator," n.d.). The planer consumed 9.2 kWh/hr, calculated from product manual specifications.

Total inputs for processing 286 BF of GLT were:

- 14.76 L gasoline (sawing)
- 74.6 kWh electricity (planing)

Inputs were divided by the mass of GLT to obtain unallocated inputs per kg (Table 14). For the reuse scenario, only sawing inputs were applied.

Table 14. Life cycle inventory inputs for waste processing salvaged mass timber panels (MTP) in the reuse and reprocess end-of-life (EOL) scenarios. The inputs are reported per kilogram (kg) of MTP.

Unallocated Input (per kg MTP)	Value	Unit	Process Description	EOL Scenario
Gasoline	0.0396	L	Sawing	Reuse, Reprocess
Electricity	0.1974	kWh	Planing	Reprocess

In addition to processing impacts, storage impacts were included, reflecting emissions from lighting, heating, and onsite transport. These inputs were taken from a study on reused softwood lumber (Bergman et al., 2013) and are shown in Table 15.

Table 15. Life cycle inventory inputs for storing salvaged mass timber panels (MTP), unallocated. The inputs are reported per kilogram (kg) of MTP, derived from Bergman et al. (2013).

Unallocated Input (per kg MTP)	Quantity	Unit	Process Description
Electricity	5.51E-03	kWh	Lighting
Propane	4.24E-04	L	Heating
Natural gas	1.73E-07	m^3	Heating
Gasoline	2.77E-03	L	Transport on-site

The inputs listed in Table 14 and Table 15 were allocated to reused and reprocessed GLT using a mass-allocation approach. Earthwise reported 25% mass loss for reused GLT during sawing and 16.67% mass loss for reprocessed GLT during sawing and planing. Offcuts were repurposed or donated, making them co-products. The mass-allocated environmental impacts of facility storage were added to the allocated impacts of sawing (and planing, for reprocess only) to produce the LCIA results.

MRF Processing

Recycled, incinerated, and landfilled C&D waste is first delivered to intermediate facilities in the waste transportation network (MRFs, transfer stations, or recyclers). Waste may arrive as source-separated or co-mingled waste. Because co-mingled waste requires additional sorting and represents a conservative estimate of processing impacts, co-mingled waste processing was used for all scenarios.

DATASMART LCI data and EPA waste distribution data were used to determine feasible EOL options and to develop assumptions about sorting, cleaning, and reprocessing requirements (LTS, 2023; EPA, 2020). All materials were compared to the list of DATASMART waste categories. When a direct match was unavailable, the “inert” category was used as the closest approximation. Inert materials are defined as non-biodegradable materials that are landfilled without additional processing.

Table 16 summarizes the material categorization and associated EOL scenarios.

Table 16. Material categorization for C&D waste materials used in the MT1 and Concrete Baseline case study buildings.

DATASMART Material Category	MT1/Concrete Baseline Material Categories	EOL Scenarios
Aluminum	Aluminum	Recycle, landfill
Concrete	Concrete	Recycle, landfill
Gypsum	Gypsum	Recycle, landfill
Inert	Carpet, glass, insulation, paint, tile, membranes	Landfill
MTP	CLT, GLT	Reprocess, reuse, recycle, incinerate, landfill
Reinforced concrete	Reinforced concrete	Recycle, landfill
Steel	Steel, rebar	Recycle, landfill

Wood, non-MTP	Lumber, plywood, wood cladding/siding	Recycle, incinerate, landfill

All materials are assumed to undergo initial processing at an MRF, and it is assumed that the material and energy inputs required would be similar for processing at a transfer station or recycler. These impacts were estimated using secondary LCI data for an MRF in Chapter 1 (Levis, 2008) (Figure 18).

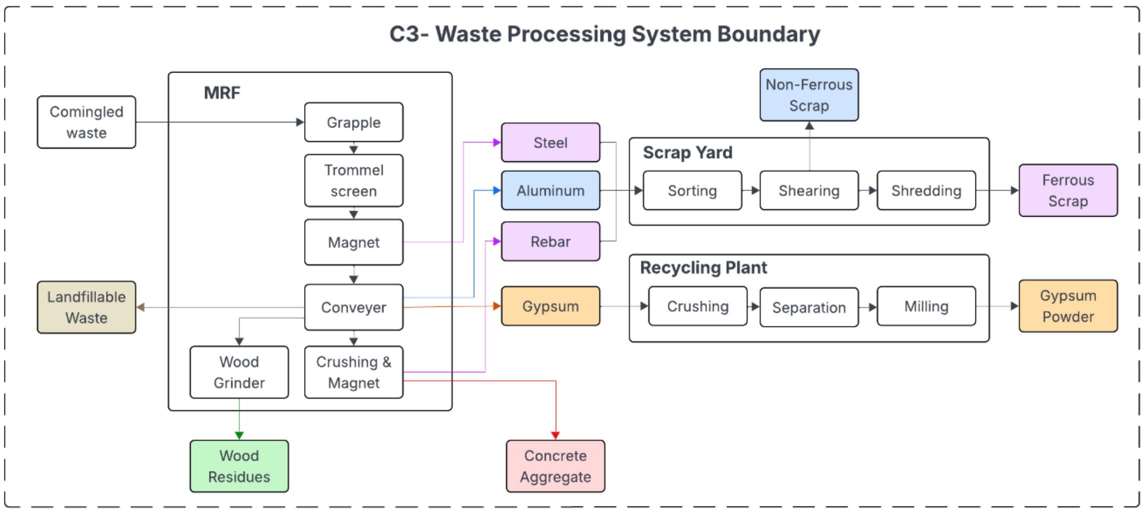

Figure 18. System boundary diagram for C3-Waste Processing activities in an MRF and other facilities. Different materials are categorized by color (green = wood; purple = ferrous metals; blue = non-ferrous metals; orange = gypsum; red = concrete; brown = inert/landfilled materials).

Some materials require additional processing after leaving the MRF, while others proceed directly to disposal (C4), at which point they exit the system boundary. For the recycle scenario, C&D waste reaches the system boundary once it leaves the intermediate facilities for manufacturing into new products.

The following subsections describe the methods used to estimate waste processing impacts for each material. The material and energy inputs for processing 1 kg of each C&D material are reported in Appendix C. All LCI inputs were modeled in SimaPro using U.S. DATASMART processes and TRACI characterization (LTS, 2023; Bare, 2012).

MTP/Wood

Waste processing impacts for recycled and incinerated MTP and other non-MTP wood waste were estimated using an LCA of a C&D MRF near Philadelphia (Levis, 2008). Reported processing steps include:

- removal of large recyclables using a grapple,
- separation of materials using a trommel screen,
- removal of ferrous metals using a magnet,
- manual sorting of wood on a conveyor, and
- chipping wood using a grinder.

The study also reports energy use for rolling stock (e.g., loaders, forklifts) and facility operations (lighting, heating, cooling). Inputs are shown in Table 17.

Table 17. Life cycle inventory inputs for waste processing 1 kilogram of waste wood in the recycle or incinerate end-of-life scenarios.

Input (per kg wood)	Quantity	Unit	Process Description
Electricity	0.00163	kWh	Facility operations
Propane	0.00087	L	Rolling stock
Electricity	0.00349	kWh	Trommel
Electricity	0.01056	kWh	Magnet
Electricity	0.00062	kWh	Conveyer (picking line)
Gasoline	0.00706	L	Wood grinder

Landfilled MTP undergoes similar processing at intermediate facilities, except that wood is not typically ground before shipment to the landfill. Therefore, landfill waste processing impacts were estimated using the same LCI as recycled and incinerated MTP, minus the grinding step.

Concrete

Levis (2008) reported the following processing activities for concrete entering an MRF:

- removal of large recyclable materials using a grapple,
- separation of larger materials from fines using a trommel screen,
- removal of ferrous metals using a magnet,
- manual sorting of concrete on the conveyor, and
- crushing with a rock crusher.

In addition to equipment-specific energy and emissions, the LCI includes rolling stock (e.g., loaders, forklifts) and facility operations (heating, lighting, cooling). Because equipment use differs from that used for wood, the share of facility operations allocated to concrete also differs.

Reinforced Concrete

Reinforced concrete follows the same general processing flow as unreinforced concrete but requires additional magnetic separation during crushing. Surveys of West Coast demolition companies indicated that jaw crushers and horizontal shaft impactors equipped with magnetic separators are used to remove rebar during crushing. Survey questions are provided in Appendix D. To reflect this additional magnetic separation, the magnetic separator input reported by Levis (2008) was doubled.

After crushing and separation, recycled concrete aggregate is typically sold for use in road base, consistent with EPA WARM assumptions. No further processing is included in the LCI for separated aggregate. Rebar, however, requires additional scrap preparation.

Rebar

After separation from concrete, rebar undergoes scrap preparation before being sold to steel manufacturers. Nucor describes the scrap preparation process as:

- sorting and separating steel from mixed scrap,

- grading by steel type,
- shearing to untangle pieces, and
- shredding into smaller pieces for transport.

Interviews with scrap yards in King County, WA provided data on equipment use and operating durations for each step. Reported equipment included front-end loaders, backhoes, Sennebogen material handlers, and industrial shears. Shredding occurs at a separate facility using an electrically powered industrial shredder.

Daily scrap throughput was divided by operating hours to estimate hourly processing rates. Fuel consumption for diesel-powered equipment was derived from construction equipment datasets or manufacturer specifications (Sattiraju, n.d.). Electricity use for shredding was calculated by dividing motor power (in kW) by hourly scrap throughput.

Prepared scrap is then sold to steel manufacturers, where it is used as feedstock in electric arc furnaces (EAFs). Rebar has an average recycled content of 98% (AISC, 2022), and external scrap accounts for 66% of EAF inputs. Melting occurs outside the system boundary and is not included as part of waste processing.

Steel

Steel follows a similar recycling pathway to rebar. In the MRF, loose steel is removed by the magnet after passing through the trommel, avoiding conveyor sorting, crushing, and magnetic separation during crushing. After arriving at the scrap yard, steel undergoes the same sorting, shearing, and shredding processes as rebar.

Aluminum

Aluminum recycling begins at the MRF, where it is separated on the picking line after removal of other materials by the grapple, trommel, and magnet (Levis, 2008). Separated aluminum is sold to scrap yards alongside steel. Interviews with scrap yards indicated that the same equipment is used to sort, stack, and compress aluminum and steel into separate piles. Long aluminum extrusions are sheared to reduce length, similar to tangled rebar. Thus, sorting and shearing inputs are identical to those used for rebar.

Gypsum

Gypsum drywall recycling is an open-loop process in which waste drywall is converted into gypsum powder for agricultural applications (EPA, 2023b). In MRFs, gypsum is removed on the picking line and sold to recycling facilities. At these facilities, gypsum is crushed and milled before being sold.

An LCA of gypsum recycling in Spain (Pedreño-Rojas et al., 2020) provides additional detail on purification steps, including crushing, separation, and milling. Calcination was excluded from this analysis because it applies only to gypsum plaster production, not agricultural gypsum powder. The LCA also includes packaging, storage, and internal transport.

Inert

Inert materials—including insulation, tile, carpet, paint, and membranes—are removed on the picking line and sent directly to landfill. Recyclable materials in poor condition are also removed at this point and landfilled. Therefore, the inert material LCI is used as the C3 waste processing profile for all landfilled materials.

C4–Waste Disposal

The final module in the end-of-life (EOL) phase is waste disposal, which applies to the incinerate and landfill scenarios. The development of C4–Waste Disposal emission factors for these scenarios is described below. As with C3–Waste Processing, this analysis builds on previous research estimating disposal impacts for MTP, using literature reviews and secondary LCA data (Bjarvin et al., 2025).

This analysis includes three separate parts:

- **MTP incineration**–determining average resin contents of MTP and modeling incineration impacts using proportional resin-to-wood ratios; validating incineration feasibility through legislative review.
- **MTP landfilling**– modeling landfill impacts for MTP using proportional resin-to-wood ratios; conducting a sensitivity analysis using other LCI data.
- **C&D waste disposal**– modeling incineration and landfill impacts for all other C&D materials using DATASMART LCI data (LTS, 2023).

Each part of this analysis is detailed in the following subsections.

MTP Incineration

Modeling the C4 impacts of incinerating MTP requires determining the weighted average resin content of U.S.-produced CLT and GLT, as well as the dominant resin types used in each product. This information was compiled from environmental product declarations (EPDs) published by U.S. manufacturers.

For GLT, an industry-wide North American EPD reported an average resin content of 1.6%, consisting of melamine formaldehyde (MF), polymer emulsion polyurethane (PEP), and phenol-resorcinol formaldehyde (PRF) (AWC & CWC, 2020). MF accounted for 0.6% of product mass, PEP for 0.5%, PRF for 0.3%, and other additives (hardeners, primers, catalysts, caustics) for 0.2%. The surveyed mills represented 44% of U.S. GLT production in 2023.

For CLT, no industry-wide EPD exists, so resin content was determined by compiling a list of U.S. CLT manufacturers using the APA Manufacturer Directory ("Manufacturer Directory," n.d.) and reviewing available EPDs. Manufacturers without valid EPDs or LCAs were excluded. Annual production capacities were obtained from manufacturer websites, enabling calculation of each facility's share of total U.S. production. Weighted average resin content was then calculated across the three manufacturers with available data (Table 18). The analysis showed that polyurethane (PUR) and MF are the predominant resin types used in U.S. CLT.

Table 18. CLT resin types and contents of U.S.-based manufacturers.

CLT Manufacturer (Location)	Annual Production Capacity (m^3)	Proportion of Total Production Capacity (%)	Resin Type	Resin Content (%)

SmartLam (Columbia Falls, MT)	52,102	15%	PUR	2.0%
SmartLam (Dothan, AL)	59,280	17%	PUR	2.0%
Mercer (Spokane, WA)	80,000	23%	PUR, MF	0.8%
Mercer (Conway, AL)	40,000	11%	PUR, MF	0.9%
Vaagen (Colville, WA)	117,000	34%	-	1.6%
Weighted Average				*1.5%*

The weighted average resin contents—1.5% for CLT and 1.6% for GLT—were used to model incineration by proportionally modeling resin and wood combustion. For example:

- CLT: 0.015 kg resin + 0.985 kg wood
- GLT: 0.016 kg resin + 0.984 kg wood

All incineration processes were modeled in SimaPro using the DATASMART LCI database and TRACI methods (LTS, 2023; Bare, 2012). Wood incineration was modeled using data from a closed MSW incinerator equipped with an electrostatic precipitator and wet flue gas scrubber. Resin incineration for PUR was modeled using the same incinerator type. Because no LCI data exists for incinerating the other resin types, PUR incineration was used as a proxy for all resin types.

To validate the assumption that MTP can be incinerated in MSW incinerators, U.S. legislation governing solid waste treatment was reviewed. The Solid Waste Disposal Act (1965), Resource Conservation and Recovery Act (1970), and Clean Air Act (1963) establish federal regulations for waste incineration, codified in Title 40 of the Code of Federal Regulations (CFR). Under:

- 40 CFR §240, C&D materials may be processed at municipal-scale incinerators (Resource Conservation and Recovery Act, 1976a).
- 40 CFR §241, resinated wood products and C&D wood waste may be classified as fuel rather than solid waste when combusted in a qualifying unit (Resource Conservation and Recovery Act, 1976b).

"Resinated wood" is defined as wood containing binders or adhesives—thus including MTP. These regulations confirm that MTP is permissible for combustion in municipal-scale incinerators under federal guidelines.

MTP Landfilling

C4–Disposal impacts for the landfill scenario were modeled in SimaPro using the DATASMART process for landfilling wood in a municipal sanitary landfill (LTS, 2023). The modeled landfill includes a base seal, leachate collection system, and municipal wastewater treatment for leachate. A DATASMART process for landfilling polyurethane resin was also used to model disposal of the resin fraction in 1 kg of CLT or GLT (LTS, 2023).

A sensitivity analysis was conducted to evaluate how landfill modeling choices influence LCIA results. The EPA WARM tool estimates GWP_{FOSSIL} for construction wood products using a mass-balance approach that tracks biogenic carbon entering the landfill and leaving as CO_2 or CH_4 (EPA, 2023a-b). These proportions are based on laboratory studies simulating landfill decomposition (Barlaz, 1998; Wang et al., 2011; Wang et al., 2013). WARM reports GWP_{FOSSIL} values of:

- 0.165 kg CO_2e/kg for lumber
- 0.055 kg CO_2e/kg for MDF

Because WARM does not include resin-specific decomposition data, landfill modeling must be applied to the wood portion only. MDF contains significantly higher resin content than MTP, which inhibits decomposition and reduces methane generation. Therefore, lumber is the more appropriate proxy for MTP in the sensitivity analysis.

C&D Waste Disposal

Waste disposal impacts for incineration and landfilling other types of C&D materials were derived from DATASMART processes for each waste type (LTS, 2023). Wood disposal impacts differ from MTP disposal because the previously-described MTP models include resin combustion. Mineral wool insulation and construction glass have dedicated landfilling processes in DATASMART, but their impacts are identical to inert materials. Because EPA reports a 100% landfill rate for both materials (EPA, 2020), they were classified as inert in this analysis.

EOL Scenario Comparison

To compare the environmental benefits and trade-offs associated with diverting C&D waste from the landfill, the cumulative waste transportation (C2), waste processing (C3), and waste disposal (C4) impacts of each material in different EOL scenarios were calculated. Because deconstruction or demolition (C1) impacts are modeled at the whole-building level, this module is excluded from the material-level comparison.

For MTP, the C2–C4 results of all five EOL scenarios are compared. For other C&D materials, the analysis compares the C2–C4 results of the recycle and landfill scenarios. This approach enables a consistent evaluation of which EOL scenarios are environmentally preferable for each material from an emissions standpoint.

Building EOL Impacts

The cumulative EOL phase (C1-C4) impacts for each case study building are calculated under different waste distribution scenarios, since the impacts of waste transportation, waste processing, and disposal vary by EOL scenario. Two waste distribution scenarios are defined for MT1, with only one applied to the Concrete Baseline: a business-as-usual (BAU) scenario and an optimistic scenario.

Waste Distribution: Business-as-Usual (BAU) Scenario

BAU waste distribution is based on EPA's national C&D waste characterization data. Aluminum and steel follow the "metals" category; concrete and reinforced concrete follow the "concrete" category; gypsum follows the "drywall" category; and all non-MTP wood follows the "wood" category. Because the EPA wood category includes mulching—an option not applicable to resin-containing engineered wood—mulching shares were reassigned to incineration.

Materials classified as inert, or materials not listed in the EPA dataset, were assigned a 100% landfill rate. Figure 19 summarizes the resulting recycle, incinerate, and landfill fractions for each material.

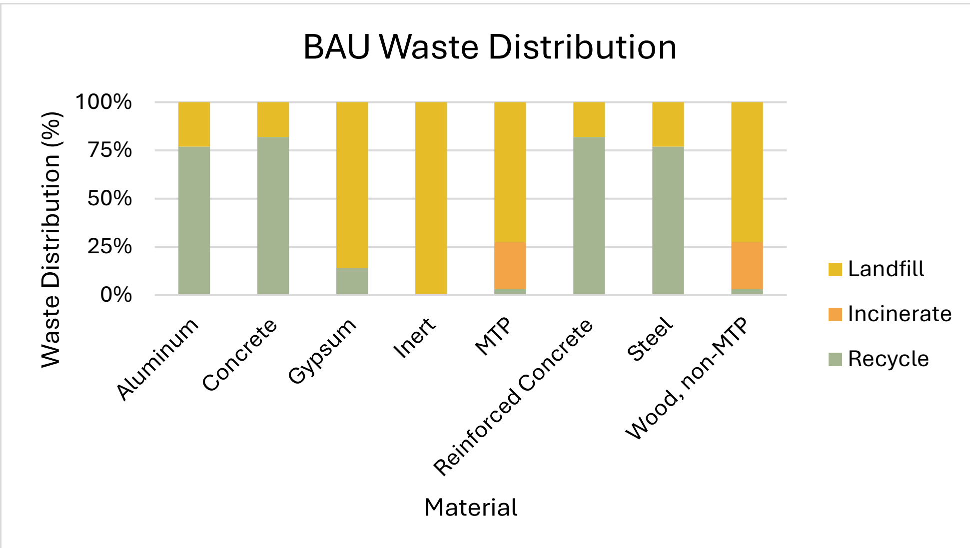

Figure 19. Waste distribution assumptions for construction and demolition (C&D) materials in two case study buildings, in the Business-as-Usual (BAU) waste scenario.

Waste Distribution: Optimistic Scenario

The Optimistic scenario modifies only the structural MTP distribution in MT1 to reflect increased recovery through selective deconstruction. All other materials, including MTP used in the enclosure, retain BAU distributions.

Recovery rates for structural MTP are taken from the mockup deconstruction in Chapter 1, where 99% of MTP (by mass) was successfully recovered for reuse and 1% was landfilled due to damage. Because the nearest reuse retailer (Whole House Building Supply & Salvage in San Mateo) sells timber without reprocessing, all recovered MTP is allocated to the reuse pathway rather than reprocessing. The remaining 1% is landfilled.

Results & Discussion

The LCIs for each module of the building life cycle were modeled in SimaPro v9.5 using processes from the DATASMART LCI database (LTS, 2023). Five impact categories were evaluated using the TRACI characterization method (Bare, 2012): fossil global warming potential (GWP_{FOSSIL}), ozone depletion potential (ODP), acidification potential (AP), eutrophication potential (EP), and smog formation potential (SFP). The LCIA results for each module within the system boundary are presented in the following subsections, beginning with C1–Deconstruction.

C1–Deconstruction/Demolition

The LCIA results of each part of the C1-Deconstruction/Demolition analysis are presented separately, starting with mockup deconstruction, followed by building demolition, and then MT1 deconstruction. The results of building demolition are presented before the MT1 deconstruction results to enable a side-by-side comparison of the impacts of demolition and deconstruction.

Mockup Deconstruction

The LCIA results for deconstructing the Borregas mockup are summarized in Table 19. A contribution analysis of individual processes shows that equipment delivery is the dominant contributor to GWP_{FOSSIL} and ODP, while the crane contributes most to the other impact categories. The boom lift, scissor lifts, forklifts, and power tools each contributed less than 8% across all impact categories. Figure 20presents the contribution analysis for each equipment type.

Table 19. Life cycle inventory analysis (LCIA) results of the Borregas mockup deconstruction analysis.

Description	GWP_{FOSSIL} (kg CO_2e)	AP (kg SO_2e)	EP (kg Ne)	ODP (kg CFC-$_{11}$e)	SFP (kg O_3e)
C1-Deconstruction	2.33E+03	2.08E+01	1.86E+00	4.36E-06	6.51E+02

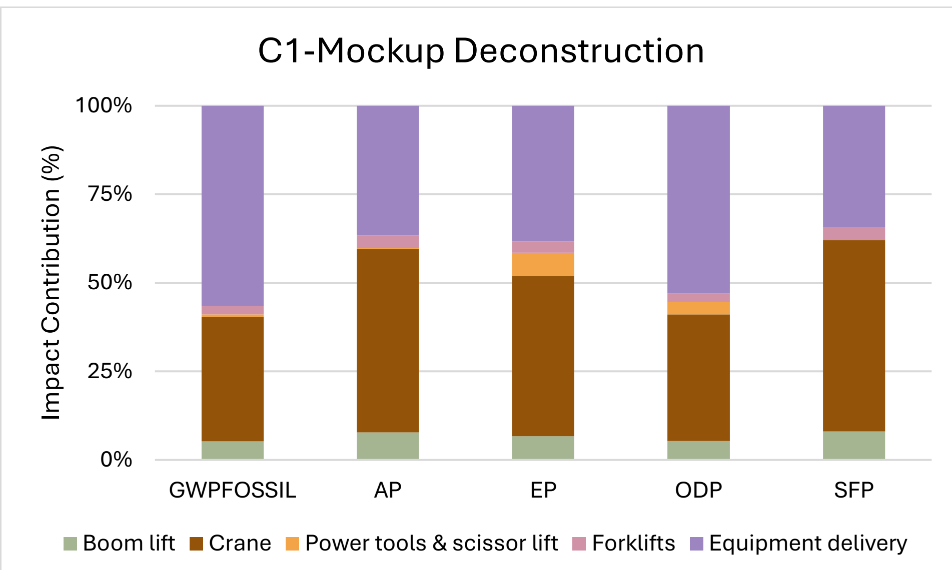

Figure 20. Contribution analysis of deconstruction processes to the overall environmental impact in five impact categories.

Sensitivity Analysis

Table 20 compares the base model to the sensitivity analysis, which replaces fuel-based equipment modeling with a machine-use-duration approach. The sensitivity model produced higher impacts in all categories except SFP, which decreased by 8%. Increases ranged from less than 1% (AP) to more than 1,500% (ODP).

Table 20. LCIA results of C1-Deconstruction for the Borregas mockup in the base case and a sensitivity analysis, where the equipment energy consumption model is changed. The percentage change in the results is also reported. A negative result denotes a decrease in impacts when the sensitivity model is applied.

Description	GWP_{FOSSIL} (kg CO_2e)	AP (kg SO_2e)	EP (kg Ne)	ODP (kg CFC-$_{11}$e)	SFP (kg O_3e)
C1-Deconstruction (Base)	2.33E+03	2.08E+01	1.86E+00	4.36E-06	6.51E+02
C1-Deconstruction (Sensitivity)	5.39E+03	2.09E+01	2.91E+00	7.18E-05	5.96E+02
% Change	131%	0%	56%	1547%	-8%

The machine-use-duration processes differ from the base model by including lubricating oil consumption, air emissions, waste treatment, and machine production. They also assume substantially higher diesel consumption rates: 7.84 kg/hour for 18.64–74.57 kW equipment and 27.63 kg/hour for >74.57 kW equipment. In contrast, the base model used equipment-specific diesel rates derived from specification sheets and manufacturer interviews (Athena, 2002). These differences in hourly diesel consumption are the primary drivers of the increased impacts in the sensitivity model.

Table 21 shows the percent change in impacts by equipment type. The boom lift and forklifts experienced decreases in most categories, while the crane showed increases across all categories. Because the crane was the largest contributor in the base model, its increased impacts dominate the overall results.

Table 21. Percent change between the sensitivity analysis of LCIA results and the base model results, by type of equipment used. A negative result denotes a decrease in impacts when the sensitivity model is applied.

Equipment Type	% Change – GWP_{FOSSIL}	% Change - AP	% Change - EP	% Change - ODP	% Change - SFP
Boom lift	-24%	-81%	-61%	582%	-83%
Crane	374%	17%	138%	4169%	2%
Power tools & scissor lift	0%	0%	0%	0%	0%
Forklifts	16%	-70%	-41%	946%	-74%
Equipment delivery	0%	0%	0%	0%	0%

Overall, the deconstruction results are highly sensitive to the equipment energy-consumption modeling approach. This underscores the need for additional primary data on equipment fuel and energy use during deconstruction, particularly for MTP buildings, as most existing LCAs rely on secondary assumptions.

Demolition (C1)

The environmental impacts of demolishing MT1 and the Concrete Baseline were modeled using five different excavator energy consumption models: Athena (per-kg basis and per-m^2 basis), Literature Review, DATASMART, and the Average model. These results are presented prior to the results of deconstruction model for MT1 to enable a comparison of demolition impacts against the deconstruction impacts.

The variation in GWP_{FOSSIL} impacts across these models is shown in Figure 21. Under the Literature Review model, demolishing MT1 has a 67% lower GWP_{FOSSIL} impact than the Concrete Baseline, and under the DATASMART model it has a 22% lower impact. In contrast, both Athena models show higher GWP_{FOSSIL} impacts for MT1: the m^2-basis model yields a 14% higher impact, and the kg-basis model yields a 194% higher impact. As a result, MT1 has a 31% higher GWP_{FOSSIL} impact than the Concrete Baseline under the Average demolition model.

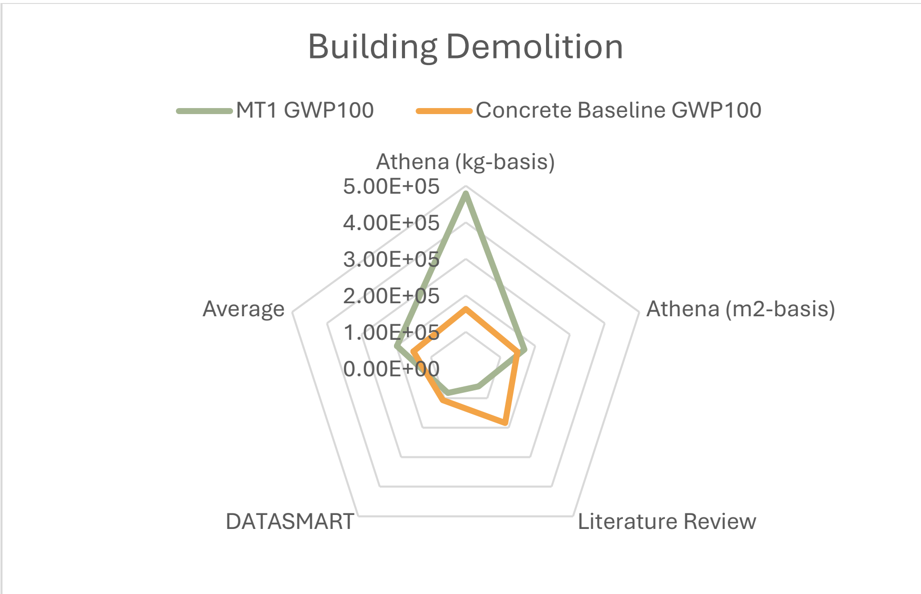

Figure 21. GWP_{FOSSIL} impacts of demolition based on five different excavator energy consumption models.

The DATASMART model is the least comprehensive of the five, as it only applies to reinforced and unreinforced concrete, rebar, and gypsum, which represent 57% of MT1's mass and 74% of the Concrete Baseline's mass (LTS, 2023). Consequently, the only difference between the two buildings under this model is the mass of each material, not the type. Because MT1 replaces reinforced concrete with MTP in the structure and enclosure, the DATASMART model does not capture potential differences in energy use when removing MTP versus reinforced concrete.

The Literature Review model has a similar limitation. Its demolition energy values are reported per usable floor area rather than gross floor area (Schenk & Amiri, 2022). Usable floor area excludes common spaces such as lobbies, corridors, structural elements, and vertical penetrations (Riscica, 2025). The authors applied a conversion factor of 0.7 to relate gross floor area to usable floor area; this factor was reversed to scale the results to a gross floor area basis, since demolition affects all building elements. However, it is unclear whether demolition energy scales linearly with floor area, as energy use may differ when demolishing structural elements versus interior partitions. Future research should quantify demolition energy per m^2 for individual assemblies, similar to how the DATASMART model provides material-specific energy consumption.

The Athena models provide the most detailed documentation of demolition equipment energy use, including energy per kilogram of material for each step of the demolition process. They are also particularly relevant because the buildings used to develop the Athena models have similar assemblies to MT1 and the Concrete Baseline. However, the Athena models are based on only one wood building and one concrete building, meaning the results are specific to those designs. This is reflected in the large differences between the kg-basis and m^2-basis results when applied to the case study buildings.

Each of the four published or database-derived models has limitations that prevent any single model from being fully representative of demolition energy for MT1 and the Concrete Baseline. Averaging the LCI inputs from the four models produces a balanced representation of demolition energy across a range of building types and designs. This is especially important because half of the models report higher energy use for wood-framed buildings, while the other half report the opposite. As such, the Average model was used in a comparative analysis against the impacts of deconstruction for MT1 in future subsections of this analysis.

Deconstruction (C1)

The environmental impacts of deconstructing MT1 were modeled by scaling up the mockup deconstruction model to a full-building level. The LCIA results are compared to demolition results for both buildings using the Average excavator energy model (Figure 22). MT1 has the highest impacts in all categories when demolished, due to its higher total diesel consumption. Deconstructing MT1 reduces diesel use and lowers impacts across all categories, consistent with findings from a study comparing deconstruction and demolition of a concrete building (Quéheille et al., 2022). Demolition of the Concrete Baseline has the lowest impacts in all categories.

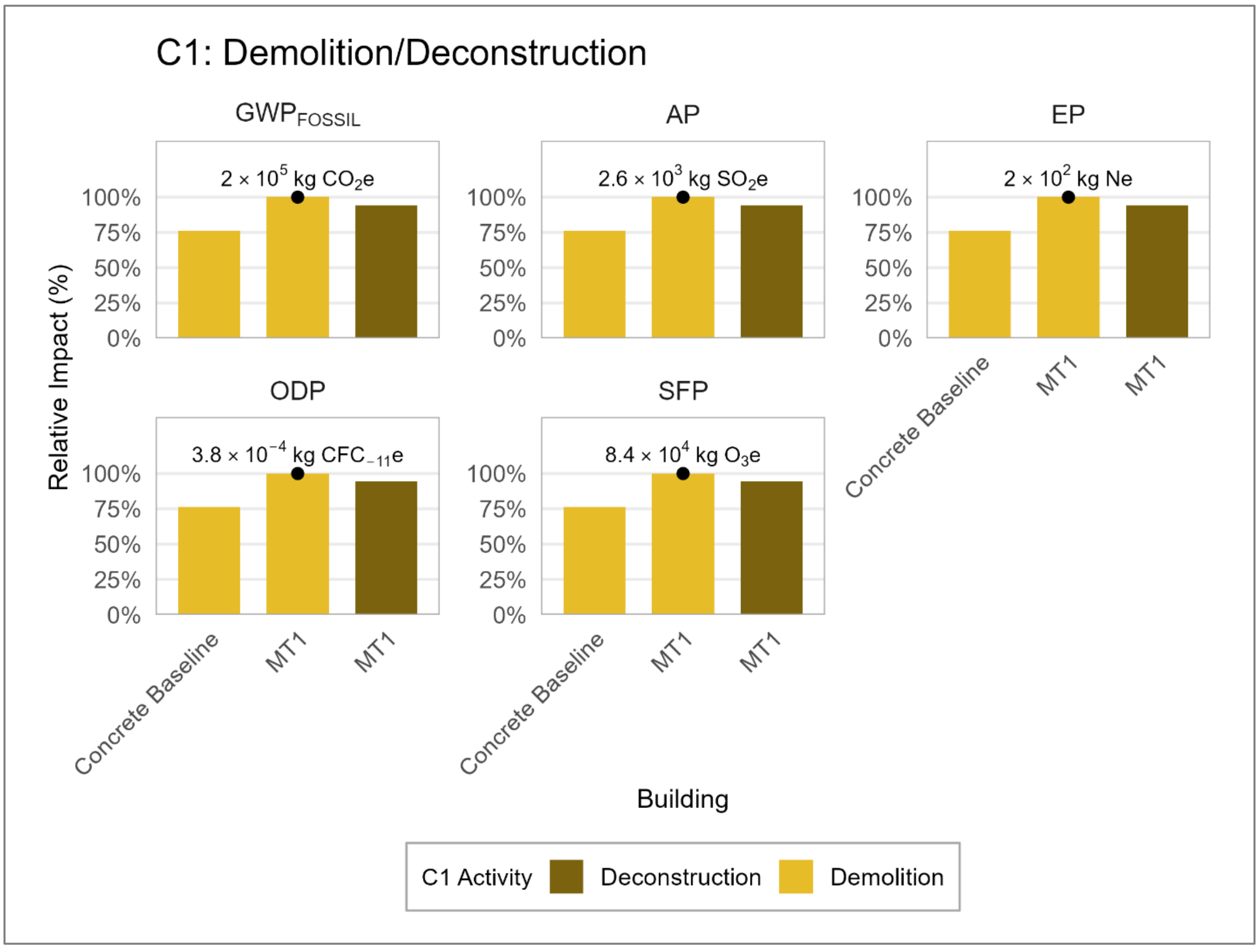

Figure 22. Relative environmental impacts of demolition or deconstruction (C1) of two case study buildings, normalized against the building/C1 activity combination with the highest impact in each impact category. Demolition was modeled using the Average excavator energy consumption model.

The LCIA results for each C1 scenario reflect total diesel consumption. Demolishing MT1 consumed the most diesel; deconstructing MT1 consumed 6% less; and demolishing the Concrete Baseline consumed 24% less than MT1's demolition. Although the Concrete Baseline has a 31% higher material mass than MT1, this was not enough to offset the higher demolition energy intensity for wood-framed structures under the Average model.

The trend of lower demolition impacts for the Concrete Baseline contrasts with the construction (A5) results, in which MT1 had lower impacts than the Concrete Baseline in all impact categories. However, the construction model applied was less comprehensive than the deconstruction or demolition models, resulting in the impacts being a direct function of the mass of materials being lifted. The influence of these contradictory trends on MT1's comparative cradle-to-grave environmental performance against the Concrete Baseline will be evaluated and discussed further in the fourth chapter of this dissertation.

C2–Waste Transportation

The LCIA results of each part of the C2-Waste Transportation analysis are presented separately, starting with the ArcGIS model for MTP, followed by the MTP waste transportation surveys, and then the waste transportation surveys for all C&D waste in the city of Sunnyvale.

MTP ArcGIS Model

The ArcGIS model estimated national average transportation distances for MTP in each EOL scenario (Table 22). The incinerate scenario had the longest average distance, while recycle had the shortest. Transportation routes for each scenario are shown in Figure 23 - Figure 26.

Table 22. Average waste transportation distances estimated by the ArcGIS model for mass timber panel (MTP) waste in five end-of-life (EOL) scenarios. The life cycle inventory analysis results are reported per kilogram (kg) of transported waste.

EOL Scenario	Distance (km)	GWP_{FOSSIL} (kg CO_2e/kg)	AP (kg SO_2e/kg)	EP (kg Ne/kg)	ODP (kg CFC-$_{11}$e/kg)	SFP (kg O_3e/kg)
Reuse/Reprocess	140.91	1.33E-02	7.38E-05	5.40E-06	2.35E-11	2.13E-03
Recycle	66.39	6.29E-03	3.48E-05	2.54E-06	1.11E-11	1.00E-03
Incinerate	292.83	2.77E-02	1.53E-04	1.12E-05	4.89E-11	4.42E-03
Landfill	116.12	1.10E-02	6.08E-05	4.45E-06	1.94E-11	1.75E-03

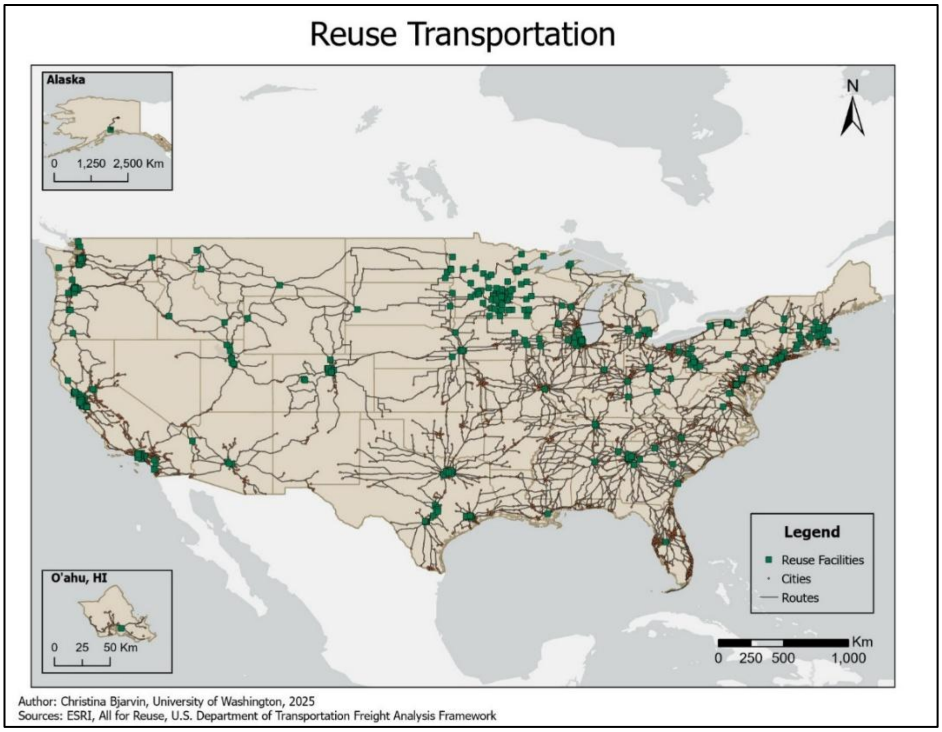

Figure 23. Waste transportation routes between cities and reus/reprocessing facilities, generated by an ArcGIS model.

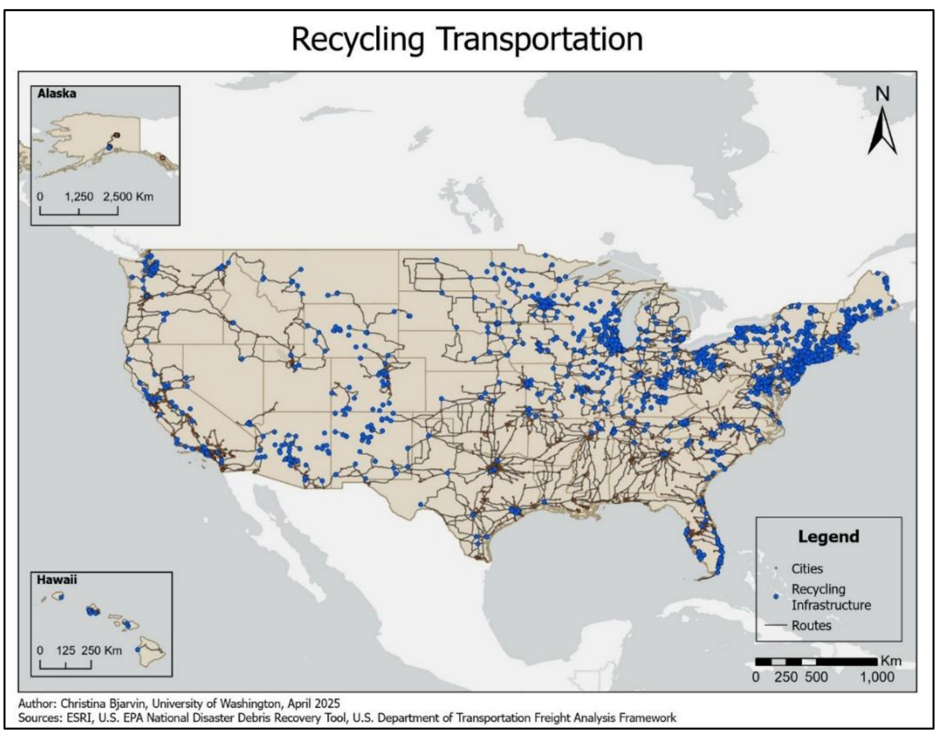

Figure 24. Waste transportation routes between cities and waste processing facilities (recycling infrastructure), generated by an ArcGIS model.

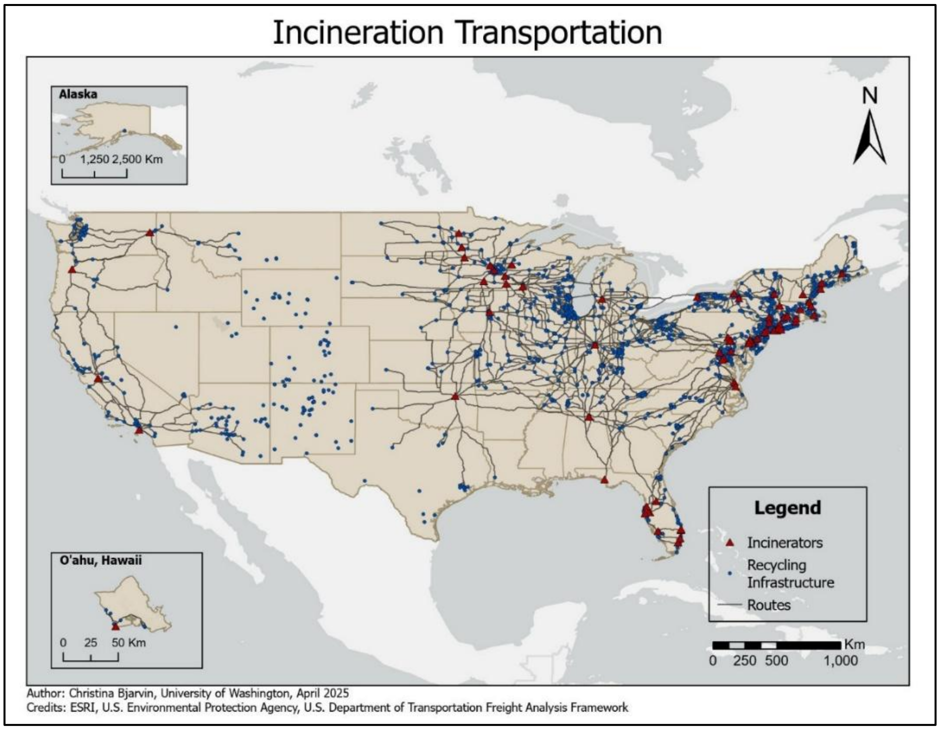

Figure 25. Waste transportation routes between waste processing facilities (recycling infrastructure) and incinerators, generated by an ArcGIS model.

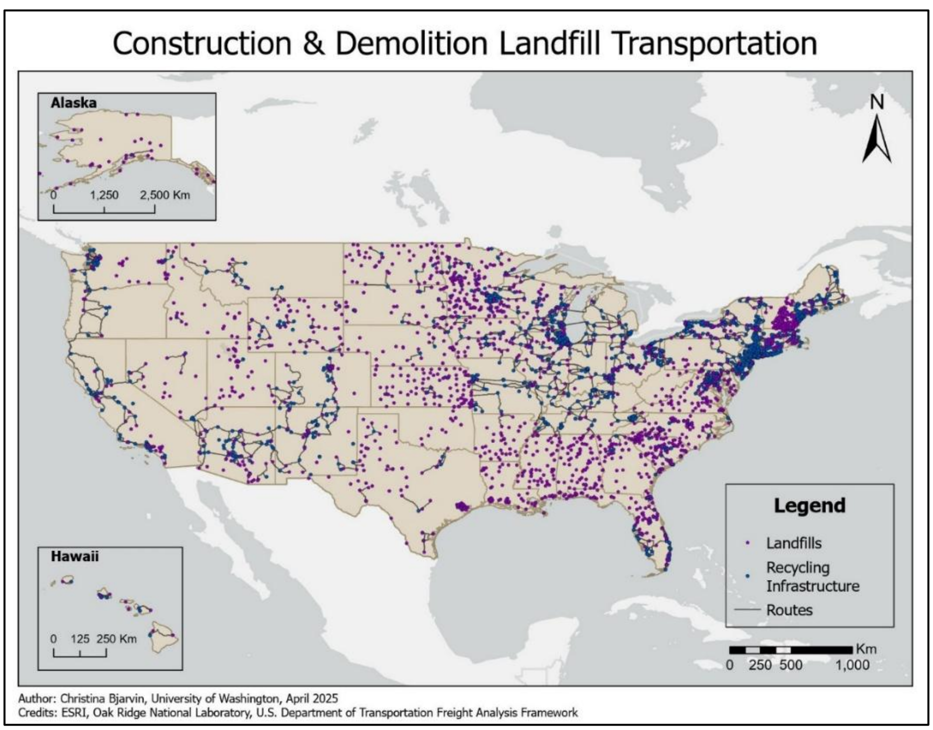

Figure 26. Waste transportation routes between waste processing facilities (recycling infrastructure) and landfills, generated by an ArcGIS model.

A sensitivity analysis was conducted by reducing the number of receiving facilities from three to one. This resulted in a 25–31% reduction in transportation distances and corresponding environmental impacts (Table 23), reflecting the linear relationship between distance and impacts in the transportation model.

Table 23. Sensitivity analysis results for the ArcGIS model. U.S. national average waste transportation distances are reported in kilometers (km). The percentage difference between the base model results and the sensitivity analysis results are also reported.

EOL Scenario	Sensitivity Distance (km)	Base Distance (km)	% Difference to Base
Reuse/Reprocess	97.74	140.91	-31%
Recycle	48.94	66.39	-26%
Incinerate	217.36	292.83	-26%
Landfill	86.59	116.12	-25%

MTP Transport Survey

To validate the ArcGIS model, surveys were conducted with C&D waste processors in five cities. Of the 83 facilities contacted, the average response rate was 69%, with the highest in Boston and the lowest in Houston (Table 24). Higher response rates generally corresponded to more transparent reporting of waste flows. The survey process also revealed substantial gaps in how cities document and disclose waste destinations. Because cities and counties are responsible for ensuring that residents have access to adequate solid waste disposal and for maintaining knowledge of where outgoing waste is ultimately sent, it is recommended that contracts with C&D processors explicitly require disclosure of final disposal locations. More comprehensive and transparent data collection by municipalities would help ensure that outgoing waste is being processed and disposed of legally and would improve the accuracy of future waste transportation assessments.

Table 24. Response rate of facilities surveyed for the MTP waste transportation analysis in five cities, with the average response rate reported (n=82).

City	Number of facilities	Response rate
Seattle	14	71%
Portland	13	62%
LA	16	69%
Boston	15	93%
Houston	24	52%
Average		69%

Figure 27 compares survey-based distances with ArcGIS estimates. On average, survey distances were lower than ArcGIS estimates for the reuse/reprocess and incinerate scenarios, higher for landfill, and similar for recycle.

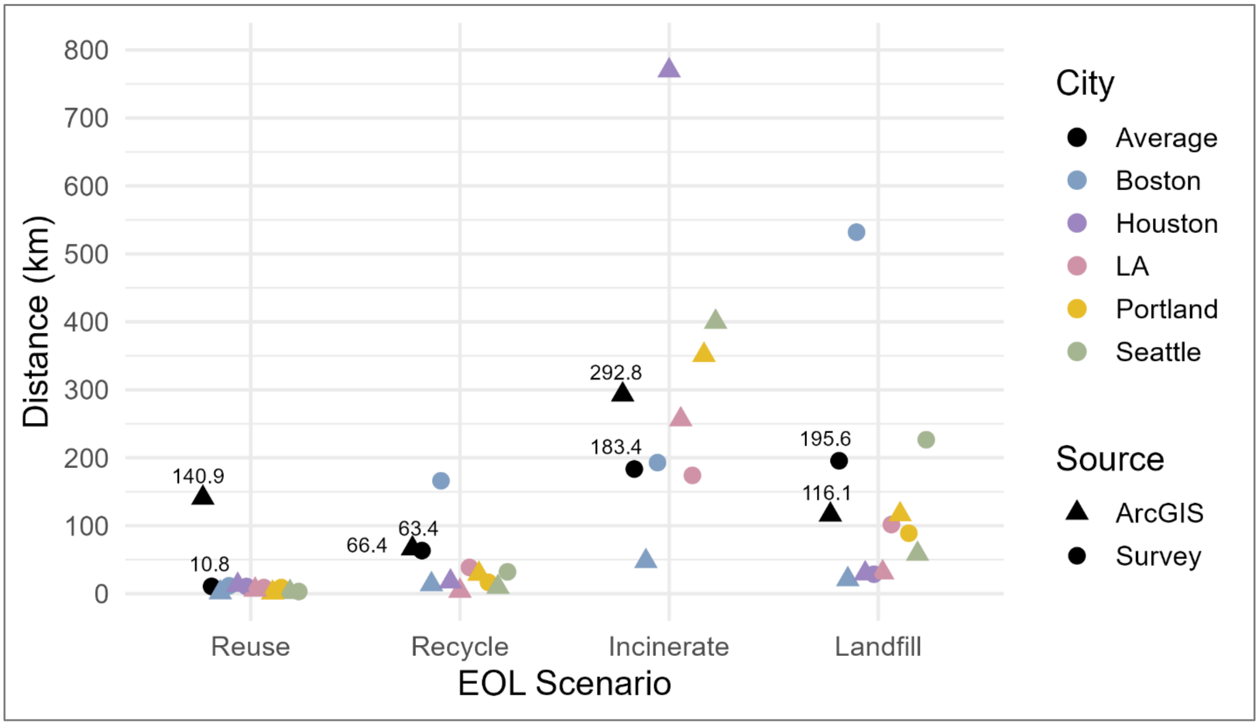

Figure 27. Average distances traveled for waste MTP in five cities and five end-of-life (EOL) scenarios using two data sources. The ArcGIS model source is represented with triangles, and the survey source is represented with circles. The national average ArcGIS distances and the calculated average survey distances for each EOL scenario are labeled.

Table 25 reports percent differences between ArcGIS and survey distances. Positive values indicate that survey distances exceeded ArcGIS estimates. The recycle scenario showed the smallest difference between ArcGIS and survey distances, likely due to more consistent recycling infrastructure across the evaluated cities. In contrast, reuse/reprocess distances varied widely, reflecting differences in local reuse capacity. The five evaluated cities generally have relatively well-developed reuse infrastructure, supported by citywide or statewide deconstruction ordinances and incentive programs, which helps keep reuse distances lower. In cities with less established reuse networks, reused material must travel farther to reach an available facility, resulting in larger discrepancies between modeled and observed distances.

Table 25. Percent differences between distances generated by the ArcGIS model and the facility survey in five cities for five end-of-life (EOL) scenarios. Negative results denote that the survey distance was lower than the ArcGIS difference, while positive results denote higher survey distances.

City	Reuse/Reprocess % Change	Recycle % Change	Incinerate % Change	Landfill % Change
Seattle	37%	233%	-	288%
Portland	58%	322%	-	186%
LA	1,364%	33%	-50%	-13%
Boston	592%	1,112%	304%	2,428%
Houston	-16%	-	-	-3%
Average	-92%	-5%	-37%	68%

In most cities, survey distances were higher than ArcGIS distances, suggesting that the ArcGIS model underestimates actual transportation distances. This is likely because the model assumes waste is sent to the three closest facilities, whereas real-world waste flows depend on contracts, tipping fees, and facility availability—not distance.

Boston exhibited the largest discrepancies. Many facilities included in the ArcGIS dataset were no longer in operation, and surveyed facilities reported sending waste to distant locations, including other U.S. states and Canada. This indicates that facility location datasets used in the ArcGIS model are outdated, particularly for incinerators and landfills.

Despite underestimating distances in individual cities, the ArcGIS model overestimated national average distances in most EOL scenarios (except landfill). This likely reflects the inclusion of rural areas, where facilities are farther apart. As a result, the ArcGIS model provides a conservative national-scale estimate, even if it performs poorly at the city scale.

A comparison with Offenhuber et al. (2012) shows that both the ArcGIS model and the survey distances exceed MSW transportation distances reported for Seattle. This suggests that C&D waste travels farther than MSW, highlighting the need for more research on differences between these waste streams.

Sunnyvale Transport Survey

This final portion of the waste transportation (C2) analysis evaluates the transport distances for C&D waste after the case study buildings are dismantled. The average distances traveled for each C&D material by EOL scenario in Sunnyvale, CA is reported in Table 26. In the recycle scenario, wood had the lowest distance of less than 10 km, as the GreenWaste Zanker Resource Recovery Facility is located close to MT1 and accepts wood for recycling, with 99.8% of their incoming wood waste being diverted from the landfill. The reuse scenario has the highest distance of the four EOL scenarios for wood, since the closest reuse center is located in a different city. Lastly, the landfill distance is relatively low, as there is a C&D landfill located in the nearby city of Milpitas.

Table 26. Average distances traveled for mass timber panels (MTP), wood, concrete, metals (steel and aluminum), gypsum, and other construction and demolition (C&D) waste in different EOL scenarios in Sunnyvale, California.

Material (EOL Scenario)	Distance (km)
MTP (reuse)	37.30
MTP/Wood (recycle)	9.70
MTP/Wood (incinerate)	18.00
Metals (recycle)	31.40
Concrete (recycle)	13.30
Gypsum (recycle)	39.70
All C&D waste (landfill)	14.60

C3–Waste Processing

The LCIA results of each part of the C3-Waste Processing analysis are presented separately, starting with reuse center processing for reused or reprocessed MTP, and followed by MRF processing for all C&D waste in the recycle, incinerate, and landfill scenarios.

Reuse Center Processing

The LCIA results for waste processing MTP in a reuse center shown in Table 27. Across all five impact categories, the reprocess scenario had higher impacts than the reuse scenario, with particularly large differences in EP and ODP.

A contribution analysis of the reprocess scenario (Figure 28) shows that planing dominates the EP and ODP impacts, while sawing contributes most to AP and SFP. Both processes contribute similarly to GWP_{FOSSIL}. Facility storage processes contribute less than 8% across all categories.

Table 27. Life cycle inventory analysis results for waste processing (C3) of mass timber panel (MTP) waste in the reuse and reprocess end-of-life (EOL) scenarios. The results are reported per kilogram (kg) of reused or reprocessed MTP.

Description	GWP_{FOSSIL} (kg CO_2e/kg)	AP (kg SO_2e/kg)	EP (kg Ne/kg)	ODP (kg CFC-$_{11}$e/kg)	SFP (kg O_3e/kg)
C3-Reprocess	1.74E-01	1.23E-03	4.68E-04	1.68E-05	3.42E-02
C3-Reuse	8.58E-02	9.11E-04	6.95E-05	1.52E-05	2.87E-02
Percent Difference	-51%	-26%	-85%	-10%	-16%

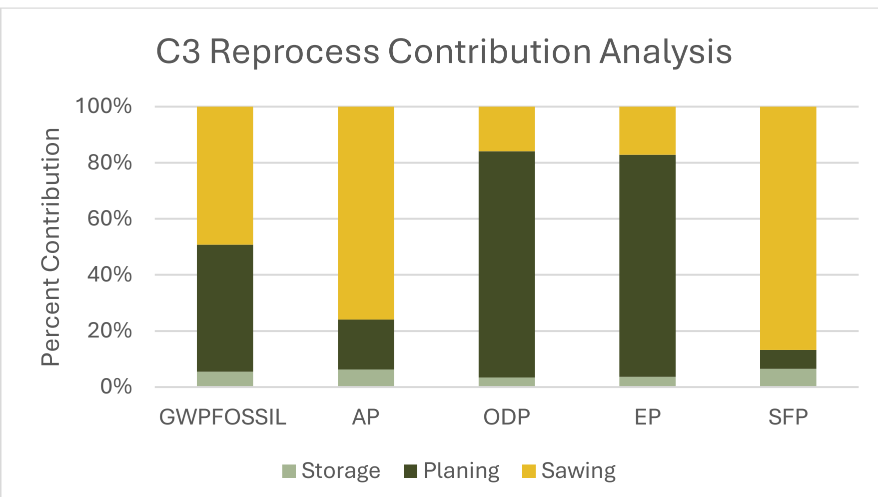

Figure 28. Contribution analysis of different processes to the overall C3-Waste Processing impacts for the reprocess end-of-life (EOL) scenario for mass timber panel (MTP) waste.

Data on waste processing energy inputs for construction wood products is limited, which constrains opportunities for validation (Table 28). Bergman et al. (2013) report a GWP_{FOSSIL} of 0.41 kg CO_2e/kg for recovering softwood framing lumber during deconstruction and storage—375% higher than C3-Reuse and 134% higher than C3-Reprocess. However, their system includes removing surface materials, extracting lumber, de-nailing, and loading, but does not include sawing or planing, making it a limited comparison for MTP waste processing.

Bjarvin et al. (2025) estimated reprocessing impacts using proxy processes for trimming virgin plywood and planing virgin lumber, without including storage. Their GWP_{FOSSIL} result was 68% lower than C3-Reprocess. This difference likely reflects the longer run times required for salvaged GLT, which must meet higher aesthetic standards for reuse as design elements. In contrast, the proxy processes represent structural wood manufacturing, where aesthetic quality is less critical and less planing is required. Even when compared to C3-Reuse, Bjarvin et al.'s (2025) GWP_{FOSSIL} is 84% lower, suggesting that proxy data for virgin wood processing does not adequately represent MTP waste processing requirements.

Table 28. GWP_{FOSSIL} results per kilogram (kg) of reused waste wood in the reprocess end-of-life (EOL) scenario, compared with other life cycle assessment (LCA) studies.

Source	GWP_{FOSSIL} (kg CO_2e/kg)	% Change to C3-Reuse	% Change to C3-Reprocess	Notes
C3-Reuse	1.14E-01	-	-	Reusing MTP (sawing)
C3-Reprocess	2.09E-01	-	-	Reprocessing MTP (sawing and planing)
Bergman et al., 2013	4.07E-01	375%	134%	Recovery of softwood lumber framing with deconstruction activities
Bjarvin et al., 2025	2.71E-02	-68%	-84%	Reprocessing MTP (proxy data for sawing and planing)

MRF Processing

The LCIA results for waste processing all C&D waste in the recycle, incinerate, and landfill scenarios at an MRF are reported in Appendix D. The results are reported per kilogram of waste; these emission factors were applied to the bill of materials in each case study building to estimate the cumulative waste processing impacts of each building, which are discussed in the "Cumulative EOL Impacts" subsection.

C4–Waste Disposal

The LCIA results of each part of the C4-Waste Disposal analysis are presented separately, starting with incinerating MTP, followed by landfilling MTP, and then incinerating and landfilling all other types of C&D waste.

MTP Incineration

The waste disposal LCIA results for MTP in the incinerate scenario are shown in

Table 29. For comparison, the impacts of incinerating 1 kg of wood waste from a boiler LCA ("Boiler #1") are included, as a portion of the residues in that study contained resins (Puettmann & Milota, 2017). Emissions data for the boiler LCA were collected from 10 softwood lumber mills and 4 plywood mills in the Pacific Northwest (PNW) and Southeast (SE). Only planer shavings from plywood mills contained resins: 10% of residues were planer shavings on a weighted average basis, and 30% of total residues originated from plywood mills. Using an average plywood resin content of 2.6% by mass, a mass-balance calculation yielded a weighted average resin content of 0.006% in the boiler residues.

This resin content was applied to the combustion model used in this study by modeling incineration of 0.99994 kg of wood and 0.00006 kg of resin ("Boiler #2"). The resulting LCIA values were then compared to the published boiler LCA results.

Table 29. Life cycle inventory analysis results for waste disposal of cross-laminated timber (CLT) and glued-laminated timber (GLT) in the incinerate end-of-life (EOL) scenario. The results are reported per kilogram (kg) of incinerated waste.

Impact Category	CLT (1.5% resin)	GLT (1.6% resin)	CLT to GLT: % Change	Boiler # 1 (Puettmann & Milota, 2017)	Boiler #2 (.006% resin)	Boiler #1 to #2: % Change
GWP_{FOSSIL} (kg CO_2e)	8.80E-02	9.04E-02	2.7%	5.88E-02	5.18E-02	-12%
AP (kg SO2e)	3.15E-04	3.17E-04	0.6%	1.04E-03	2.91E-04	-72%
EP (kg Ne/kg)	7.48E-04	7.49E-04	0.2%	2.13E-03	7.30E-04	-66%
ODP (kg CFC-$_{11}$e/kg)	8.93E-10	8.97E-10	0.4%	3.01E-09	8.33E-10	-72%
SFP (kg O_3e/kg)	9.92E-03	9.97E-03	0.5%	3.32E-02	9.18E-03	-72%

Increasing the resin content of an incinerated wood product increases impacts across all categories, as shown by the higher values for GLT relative to CLT. The impacts of incinerating residues with a 0.006% resin content were lower than those for both MTP products. Comparing the two residue-incineration models shows that the combustion model used in this study (Boiler #2) underestimates impacts relative to the published boiler LCA (Boiler #1). Puettmann and Milota (2017) noted that their GWP_{FOSSIL} results were higher than those in the USLCI boiler process because the USLCI model did not include auxiliary material inputs, while their results for other impact categories were lower due to lower reported air emissions.

In contrast, the boiler LCA results were higher than the DATASMART incineration model in all impact categories. This difference is unlikely to be caused by resin combustion in the DATASMART model, as incinerating 1 kg of wood using the DATASMART process produced identical results across all categories. The differences are therefore more likely due to variations in auxiliary material inputs and air-emission values between the two models.

Because empirical data on the environmental impacts of incinerating wood products with higher resin contents is limited, it remains uncertain whether the DATASMART incineration model accurately represents resinated wood products or whether using PUR as a proxy for other resin types is appropriate. Future work should incorporate incineration processes for additional resin types as new LCI data becomes available.

MTP Landfilling

The waste disposal LCIA results for CLT and GLT in the landfill scenario are shown in

Table 30. Unlike the incineration results, the impacts of landfilling CLT and GLT are identical. When biogenic materials such as wood enter the anaerobic environment of a landfill, a portion of the biogenic carbon is released as CO_2, some as CH_4, some as volatile organic compounds (VOCs) or leachate, and the remainder is stored permanently (EPA, 2023a). The proportion of carbon released as CH_4 drives most of the resulting GWP_{FOSSIL} impact, and the amount and timing of CH_4 emissions depend on factors such as moisture content and product composition (EPA, 2023a).

Table 30. LCIA results for waste disposal of cross-laminated timber (CLT) and glued-laminated timber (GLT) in the landfill end-of-life (EOL) scenario. The results are reported per kilogram (kg) of landfilled waste.

Impact Category	CLT (1.5% resin)	GLT (1.6% resin)
GWP_{FOSSIL} (kg CO_2e)	1.58E-01	1.58E-01
AP (kg SO2e)	9.27E-05	9.27E-05
EP (kg Ne/kg)	7.12E-03	7.11E-03
ODP (kg CFC-$_{11}$e/kg)	4.07E-09	4.07E-09
SFP (kg O_3e/kg)	2.52E-03	2.52E-03

Wood products with higher wood content (and lower resin content) can have higher GWP_{FOSSIL} impacts if more wood is available to decompose. Conversely, higher resin contents can inhibit methanogenic bacteria and reduce CH_4 generation. The small difference in resin content between CLT and GLT is insufficient to produce meaningful differences in CH_4 emissions under the modeled conditions.

The DATASMART landfill model used in this analysis represents a reactive organic landfill with landfill gas (LFG) and leachate recovery systems, based on Swiss landfill technology circa 2000 (LTS, 2023). To assess the influence of landfill type on CH_4 generation and GWP_{FOSSIL}, a sensitivity analysis was conducted using the EPA WARM landfill model. The results are shown in .

Table 31.

Table 31. GWP_{FOSSIL} results per kilogram (kg) of landfilled mass timber panel (MTP) waste in the landfill end-of-life (EOL) scenario, compared with an alternate landfilling model under a sensitivity analysis.

Impact Category	DATASMART landfill model (base)	EPA WARM landfill model (sensitivity)	Percent Difference
GWP_{FOSSIL} (kg CO2e/kg wood)	1.58E-01	1.65E-01	4.4%

Differences between the two models can be attributed to LFG collection assumptions. WARM provides emission factors for three LFG management types: flaring, electricity generation, and no LFG recovery. The national average emission factor reported by WARM is identical to the "no LFG recovery" scenario, indicating that most U.S. landfills do not employ LFG recovery. Under the Resource Conservation and Recovery Act, Subtitle D landfills must monitor CH_4 emissions during operation and for 30 years after closure, and landfills constructed after 2014 with a design capacity greater than 2.5 million metric tons must collect and control LFG (Resource Conservation and Recovery Act, 1976c-d). Older landfills, however, are exempt from these requirements.

Because the WARM national average reflects limited LFG recovery, its GWP_{FOSSIL} value is higher than that of the DATASMART model, which assumes gas collection. While WARM is more representative of U.S. landfill conditions, it only provides emission factors for GWP_{FOSSIL}. Until WARM includes additional impact categories, the DATASMART landfill model remains the more complete option for multi-category LCIA. Moreover, the difference between the two models is small, suggesting that the Swiss-based DATASMART model is representative of U.S. landfill practices.

C&D Waste Disposal

The LCIA results for disposing of all C&D waste in the incinerate and landfill scenarios are reported in Appendix D. The results are reported per kilogram of waste; these emission factors were applied to the bill of materials in each case study building to estimate the cumulative waste disposal impacts of each building, which are discussed in the "Cumulative EOL Impacts" subsection.

EOL Scenario Comparison

In this comparison, the cumulative impacts of waste transportation (C2), waste processing (C3), and disposal (C4) are calculated for different C&D materials in each EOL scenario. First, the C2-C4 impacts of MTP in five EOL scenarios are compared, followed by a comparison of the recycle and landfill scenario impacts for all C&D materials.

MTP EOL Comparison

The comparative EOL impacts for MTP were evaluated in five EOL scenarios: reuse, reprocess, recycle, incinerate, and landfill (Figure 29). In the EOL phase, reusing MTP yielded the highest global warming, acidification, and smog formation impacts, resulting from the use of fossil-based electricity and diesel fuel during waste processing (planing and sanding). Landfilling MTP yielded the highest impacts in the remaining impact categories, since nitrogen-containing leachate is generated during landfill degradation, causing eutrophication. Leachate treatment at wastewater processing facilities emits ozone-depleting substances during combustion, causing ozone depletion.

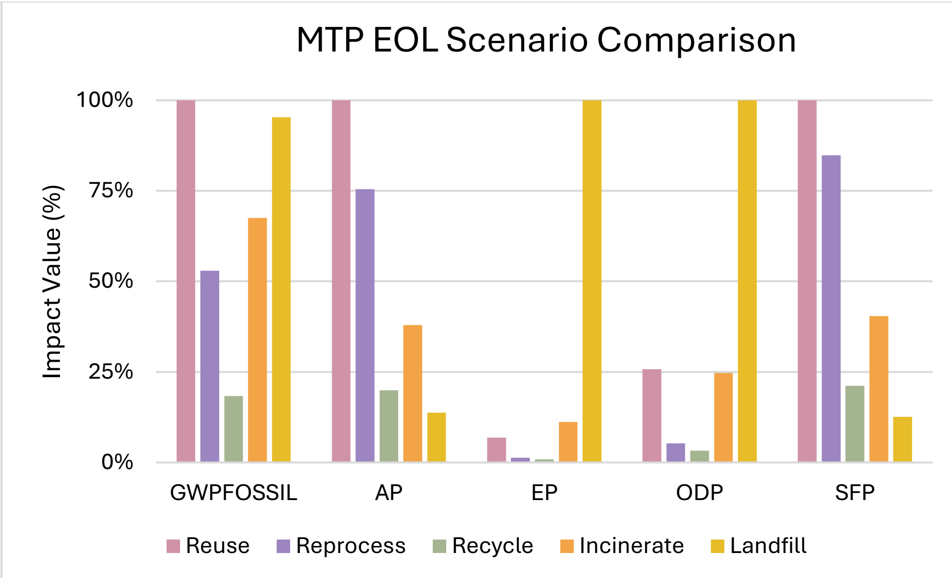

Figure 29. Relative life cycle inventory results of the end-of-life (C2-C4) phase for mass timber panels (MTP) in five end-of-life (EOL) scenarios, per kilogram of MTP. Results are normalized against the EOL scenario with the largest impact in a given impact category.

Conversely, the recycle scenario for MTP had the lowest impacts in all five impact categories. Recycling MTP consumed less diesel during waste processing compared to the reuse and reprocess EOL scenarios, and avoids the disposal burdens related to incineration and landfilling.

Recycle and Landfill Comparison

The C2-C4 impacts of recycling 1 kilogram and landfilling of each C&D material were evaluated and compared. The resulting percent change in impacts from recycling instead of landfilling is shown in Figure 30. Recycling C&D materials reduces the EOL impacts of most materials across the five impact categories, with some exceptions for wood products and gypsum.

MTP and other wood products (represented in brown) have reductions in most impact categories, since the degradation of wood in landfills releases methane emissions, causing global warming. Nitrogen in wood is also released in leachate, causing eutrophication. Treatment of leachate via combustion also causes ozone depletion[2]. However, recycling MTP and wood increases AP and SFP impacts, since gasoline is consumed during wood chipping, a process that is avoided during landfilling. Gasoline combustion emits sulfur dioxide (SO_2) and nitrous oxides (NO_x), both of which contribute to acidification and smog formation. While recycling wood products instead of landfilling them avoids environmental burdens caused by landfill degradation, the use of fossil gasoline and diesel during recycling processes inhibits recycling from having a better performance in several air pollution categories.

Gypsum has reductions in most impact categories when recycled, with the exception of global warming and eutrophication. Gypsum has a notably large reduction in AP impacts, since sulfate in landfilled gypsum is converted to hydrogen sulfide (H_2S) by sulfate-reducing bacteria. Combusting

[2] Since all C&D waste has leachate emissions when landfilled, ODP is the only impact category in which recycling has lower impacts than landfilling for every material.

H_2S during biogas treatment forms SO_2, causing acidification (Knobloch, 2024). However, recycled gypsum has higher GWP_{FOSSIL} and EP impacts than landfilled gypsum, due to the use of fossil-powered electricity during crushing in the recycling process. Fossil electricity provided by the average U.S. grid uses coal, which releases upstream phosphate emissions during the mining process.

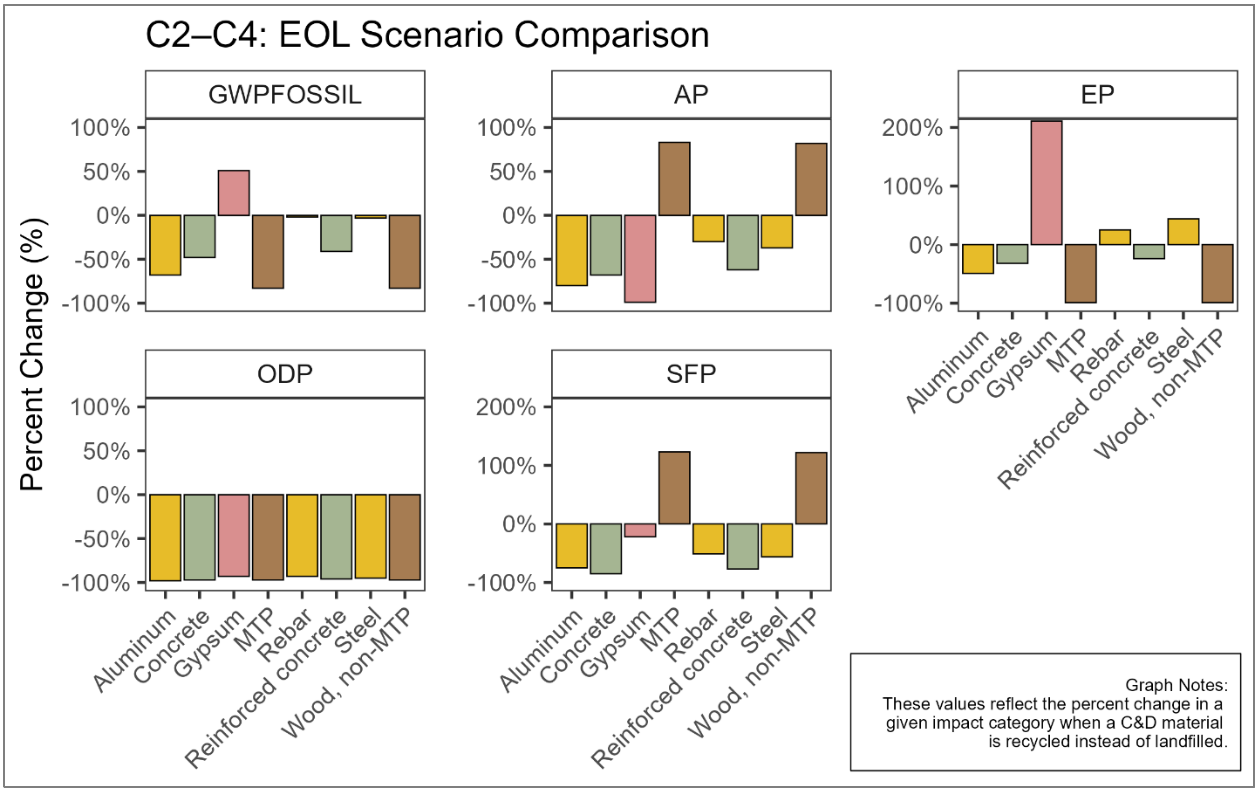

Figure 30. Percentage change in end-of-life (EOL) environmental impacts when a C&D material is recycled instead of landfilled. Metals are yellow, gypsum is pink, wood products are brown, and concrete is green.

In sum, recycling MTP results in the lowest environmental impacts among the five considered EOL scenarios for MTP, whereas the reuse and landfill scenarios had the highest impacts. Recycling other types of C&D waste has similar advantages compared to the landfill scenario in most impact categories, with some exceptions for materials that consume large amounts of fossil-based electricity or fuels during recycling processes. Switching to renewable energy during waste processing could reverse this trend, since the use of fossil energy drives the majority of the reuse and recycling EOL impacts. Moreover, considering the benefits associated with virgin material substitution could further highlight the environmental benefits of diverting waste from landfills, which will be evaluated in the third chapter of this dissertation.

Building EOL Impacts

The cumulative impacts of the EOL phase for each case study building were evaluated under a BAU scenario, in which each building was demolished, and conventional waste distribution methods were assumed. The cumulative impacts of a second Optimistic scenario were evaluated for MT1, in which the building was deconstructed to recover all structural MTP for reuse.

To demonstrate the influence of individual materials to the EOL impacts, the waste transportation (C2), waste processing (C3), and waste disposal (C4) impacts were first assessed without the deconstruction or demolition (C1) results. The material-level contributions to the C2-C4 impacts in each waste distribution scenario are shown in Figure 31. MT1 had higher C2–C4 impacts than the Concrete Baseline in all categories, regardless of the waste scenario applied. The BAU scenario produced the highest GWP_{FOSSIL}, EP, and ODP impacts, while the Optimistic scenario produced the highest AP and SFP impacts, driven primarily by the reuse of MTP.

For GWP_{FOSSIL} and EP, landfilling MTP in the MT1 BAU scenario generated relatively high impacts due to CH_4 emissions and leaching of nitrogen-containing compounds during anaerobic decomposition. Diverting MTP to reuse in the Optimistic scenario reduced both impacts. The Concrete Baseline had lower impacts because it contains far less wood.

For ODP, the MT1 BAU scenario also had higher impacts than the Optimistic scenario because landfilling MTP includes the burdens of incinerating leachate sludge with crude oil and coal, which emits ozone-depleting substances. The Concrete Baseline BAU scenario has lower ODP impacts because more of its materials are recycled rather than landfilled. Under the Optimistic scenario, MT1 landfills less material than the Concrete Baseline, resulting in lower ODP impacts.

For AP and SFP, the MT1 Optimistic scenario has the highest impacts due to diesel consumption during MTP reuse processing. Diesel combustion emits NO_x and SO_2, contributing to acidification and smog formation. Under the BAU scenario, AP and SFP impacts are similar between the two buildings, with MTP and reinforced concrete contributing most to MT1 and the Concrete Baseline, respectively. This suggests that similar amounts of diesel are consumed when MTP is landfilled and reinforced concrete is recycled on a functional-equivalence basis, since the amount of MTP landfilled (~75%) is similar to the amount of concrete landfilled (~80%) under the BAU scenario.

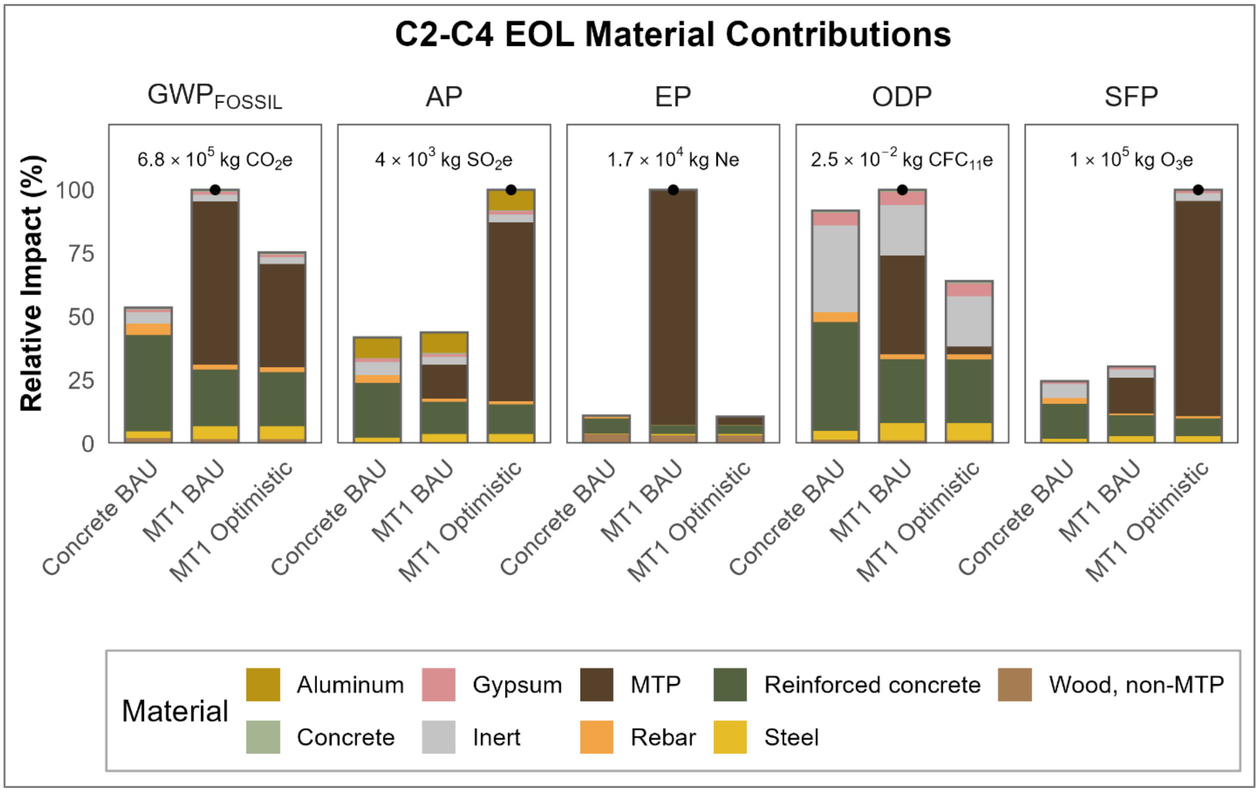

Figure 31. Relative end-of-life (EOL, C1-C4) environmental impacts for the MT1 and Concrete Baseline case study buildings under different waste scenarios, normalized against the building/waste scenario combination with the highest impact in each impact category. The relative contribution of each waste material to the entire impact is also shown.

A contribution analysis of total C1–C4 impacts shows that C1 (demolition or deconstruction) contributes most to AP and SFP, while C3/C4 (waste processing and disposal) contributes most to the other categories (Figure 32). Diesel-intensive equipment such as excavators, skid steers, and cranes drive AP and SFP impacts. Landfilling MTP and wood drives waste processing and disposal GWP_{FOSSIL} and EP impacts due to CH_4 emissions and nitrogen leaching. An exception is the GWP_{FOSSIL} impact of the Concrete Baseline, where C1 dominates because the building contains 26 times less wood than MT1, making landfill-related CH_4 emissions negligible relative to fossil fuel combustion during demolition. High ODP contributions from waste disposal arise from sludge incineration during landfill leachate treatment.

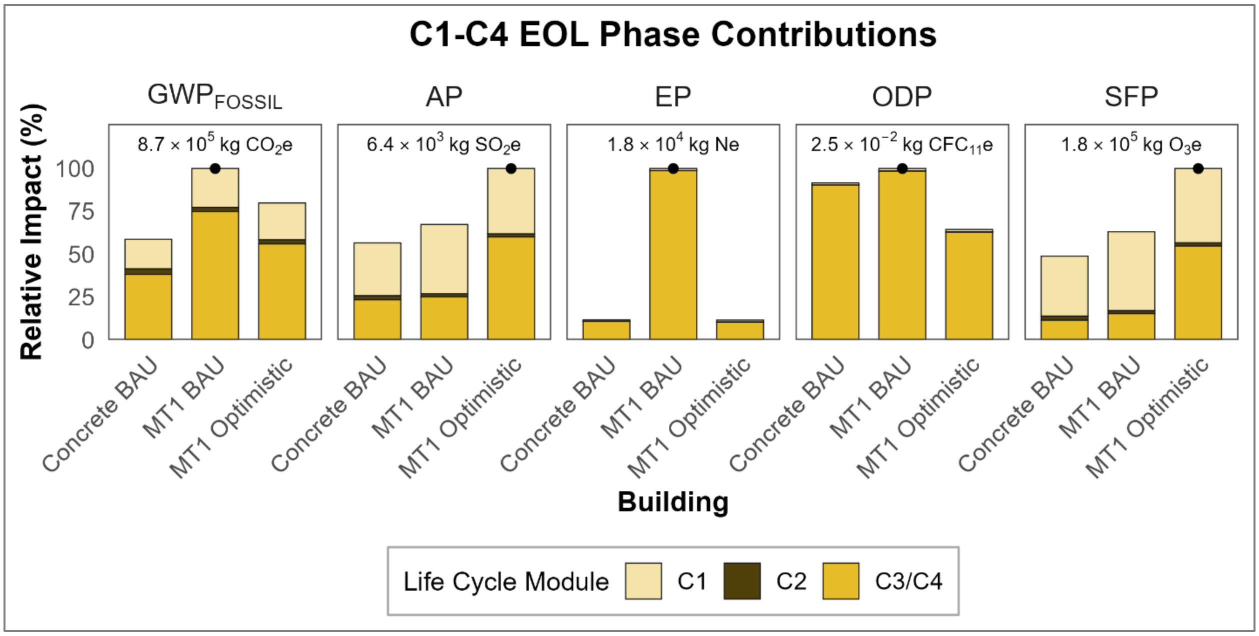

Figure 32. Relative C1-C4 environmental impacts for the MT1 and Concrete Baseline case study buildings under different waste scenarios, normalized against the building/waste scenario combination with the highest impact in each impact category. The relative contribution of each life cycle phase to the entire impact is also shown.

Comparing MT1's BAU and Optimistic scenarios shows that deconstructing the building to recover MTP for reuse reduces GWP_{FOSSIL}, EP, and ODP impacts. These reductions are driven by avoiding landfill emissions. The reduction in ODP is smaller because MTP contributes less to ODP than to GWP_{FOSSIL} or EP. However, AP and SFP increase in the Optimistic scenario due to additional diesel consumption during MTP reuse processing.

In summary, the majority of the environmental impacts produced in the EOL phase of the case study buildings come from disposal of landfilled materials, particularly MTP and other wood products. Landfill degradation of biogenic wood emits methane and nitrogen-containing leachate, resulting in global warming and eutrophication. Leachate treatment also contributes to ODP impacts, though this is applicable to other C&D waste besides MTP and other wood products. Conversely, demolition or deconstruction has a larger contribution than disposal to acidification and smog formation, due to the use of diesel-powered dismantling equipment.

Comparing the EOL impacts of both buildings under different waste distribution scenarios reveals an environmental advantage for the Concrete Baseline in most impact categories, with the exception of EP and ODP. These two impacts are driven primarily by landfilling processes, so reducing the amount of landfilled MTP in the Optimistic scenario results in MT1 having similar or lower impacts. Diverting MTP to reuse reduces the cumulative EOL phase impacts of the MT1 building in all impact categories except for acidification and smog formation, since reusing MTP avoids landfill-related burdens but uses more diesel during waste processing. While deconstructing MT1 in the Optimistic scenario used less diesel compared to demolition in the BAU scenario, the additional diesel used during reuse waste processing outweighed this benefit, resulting in higher AP and SFP results when MT1 was deconstructed to recover MTP for reuse.

Limitations

Assumptions made during each module of the EOL life cycle introduce uncertainty into the conclusions drawn from this analysis, including building demolition or deconstruction (C1), waste transportation (C2), waste processing (C3), and disposal (C4). The limitations presented in module are discussed separately, starting with C1.

Deconstruction/Demolition

In the C1-Deconstruction/Demolition module, the impacts of deconstructing MT1 were based on a hand deconstruction model applied to a mass timber mockup structure. The mockup lacks key MT1 building elements, such as interior/exterior walls, ceilings, and interior assemblies. Uncertainty remains about whether fuel consumption models based on material mass and assembly height apply to these missing elements.

The deconstruction model applied to MT1 involves separating the concrete topping from the MTP decking system to expose structural MTP, which is then removed using impact drivers to minimize damage. This model is based on the methods used to deconstruct a mockup structure of MT1. However, the mockup deconstruction team noted that this “hand deconstruction” approach is not feasible for full-scale buildings. Therefore, the purpose of the deconstruction analysis is to illustrate the potential environmental benefits of reusing MTP rather than landfilling it.

The demolition model estimated excavator energy consumption using four different models: the Athena models (m^2-basis and kg-basis), the Literature Review model (Schenk & Amiri, 2022), and the DATASMART model (LTS, 2023). Each model has inherent limitations. The Athena models are based on only one timber building and one concrete building, making their results highly specific to those designs. The Literature Review model reports demolition energy per usable floor area, a functional unit that does not scale directly to the case study buildings. The DATASMART model provides material-specific energy consumption values but only for four materials present in the buildings. To address these limitations, an Average model was created by averaging the energy consumption values from all four models, and this average was used to estimate excavator demolition impacts. Although this approach reduces uncertainty compared to selecting a single model, a weighted average would have been preferable to give more influence to models with larger sample sizes. This was not possible because the DATASMART model does not document how its data were collected. Future research should quantify the relationship between excavator demolition energy consumption and specific materials on a mass basis.

Waste Transportation

In the C2-Waste Transportation module, a geospatial model was created in ArcGIS to generate transportation distances between different wood waste processing facilities. It is unclear how many waste processing facilities each city uses, or how many disposal facilities each processor contracts with. The model assumed three destinations per facility type to avoid underestimating distances, but a sensitivity analysis showed that changing this number alters impacts by 25–31%.

Data used in the ArcGIS model also had issues with data quality and completeness. Within the recycling infrastructure dataset, it was not possible to distinguish recyclers from waste processing facilities such as MRFs or transfer stations, so the recycle scenario treated transfer stations and

MRFs as final destinations, assuming negligible transport between them—an assumption based on municipal solid waste, not C&D wood. Similarly, the incinerator facility dataset excludes industrial biomass users (e.g., lumber mills, cement kilns), likely underestimating facility counts and overestimating distances. Lastly, the landfill facility dataset lacks capacity and tipping-fee information, which can strongly influence contracting decisions—basing a future geospatial model on these parameters rather than proximity may yield a better fit to the results of the waste transportation surveys conducted in five cities. These gaps in the ArcGIS datasets likely reflect delays in government data processing and release, as this process can take several months or years.

The limitations associated with the ArcGIS model were addressed by surveying actors in the waste transportation network in five cities, which relied on the use of interviews and public records requests to collect data. The results of the surveys depend on facility transparency; some LA facilities declined participation, potentially due to recent illegal C&D fines dumping reports (Margolis, 2025), while other facilities withheld proprietary partner information. Public records requests produced complete data only in Boston, whereas other cities provided heavily redacted documents. In sum, the responses provided by the surveyed facilities may not reflect actual practices, in an effort to protect their proprietary information or cover any illegal disposal practices.

Waste Processing

In the C3-Waste Processing module, primary LCI data collection for reused and reprocessed MTP introduces uncertainty. Data were collected from a single batch of reprocessed GLT rather than annual averages at a reuse center, and only one facility was sampled due to time constraints. Future studies should aim to collect annual processing data from multiple facilities to validate the reuse and reprocessing models for MTP.

The LCI data for C&D waste in MRFs came from a single Pennsylvania MRF nearly 20 years ago. It is unknown whether these data represent current national MRF practices, as this type of data has never been collected on a national scale and made publicly-available. Moreover, due to a lack of published literature on U.S.-based gypsum recycling, LCI inputs for gypsum recycling were based on a Spanish recycler. The representativeness of Spanish gypsum recycling infrastructure to U.S.-based practices is unclear.

The waste processing global warming potential (GWP_{FOSSIL}) impact for concrete in this analysis was 77 to 83% higher than three other published studies in the U.S. (Wilburn & Goonan, 1998; Cochran, 2006; McIntyre et al., 2009). However, the waste processing data for two of the concrete LCA studies was based on data collected in 1998, which is more outdated than the study used to model waste processing impacts for this dissertation.

Similarly, the waste processing GWP_{FOSSIL} impact for steel was also 72% higher than a DATASMART process based on Swiss infrastructure (LTS, 2023). While Levis' (2008) study was deemed to have the best temporal and geographic data quality, it is unknown whether the studied MRF's processing technology is representative of current U.S. waste processing infrastructure. Additionally, the waste processing GWP_{FOSSIL} impact for wood and gypsum in this dissertation were 85% and 93% higher than an estimate of diesel consumption by machinery used for each material during waste processing (Cochran, 2006; EPA, 2021). Nevertheless, the C3-Waste Processing impacts for C&D

waste in this dissertation represent a conservative estimate of the material and energy inputs required compared to published literature.

Waste Disposal

In the C4-Waste Disposal module, MTP incineration was modeled based on a resin-to-wood ratio. Polyurethane (PUR) incineration was used as a proxy for all resin types, since this is the only resin with available LCI data. CLT predominantly uses PUR, making the model reasonably representative for this product, but GLT commonly uses other resins for which no combustion LCIs exist. Using PUR as a proxy may misrepresent GLT incineration impacts, but it remains the best available option.

In this research, landfilling MTP generated disproportionately high EP impacts relative to other waste types, leading to the conclusion that BAU disposal practices for waste MTP would result in higher EP impacts for MT1 compared to the Concrete Baseline building. These elevated impacts stemmed from the high biological oxygen demand, chemical oxygen demand, and ammonium emissions to water associated with short-term leachate treatment, as well as the incineration of sludge produced during that treatment. However, the background LCI data used to model landfilling of MTP is based on average Swiss landfill technology, which does not capture variation in factors influencing leachate formation—such as waste moisture content or degree of decomposition (EPA, 2023a).

The LCI data for landfilling MTP was sourced from the DATASMART LCI database (LTS, 2023), a U.S.-specific dataset created by combining and modifying LCI data from the USLCI and ecoinvent databases. The landfill process used in this dissertation originates from ecoinvent and therefore reflects European technology and market conditions. Specifically, the modeled landfill represents Swiss sanitary landfill technology from 2000, characterized by a base seal, landfill gas and leachate collection systems, and downstream leachate treatment at a wastewater treatment plant. Sanitary landfills commonly also include barrier caps, landfill gas flare stations, and soil covers (Nanda & Berruti, 2020), though the documentation does not clarify whether these features are included in the Swiss model. Reliance on average technology data limits the ability to account for variation in landfill conditions, including age, resource recovery technologies, and wastewater treatment practices.

The representativeness of early-2000s Swiss landfill technology for modern U.S. landfills is uncertain, particularly given differences in regulatory frameworks. Rather than imposing increasingly stringent environmental controls on landfills, Switzerland prohibited combustible construction waste from landfills beginning in 2000 (Systematic Collection of Laws, 2015). In contrast, construction wood in the U.S. may be disposed of in either municipal solid waste (MSW) landfills or construction and demolition (C&D) landfills. U.S. MSW landfills operate under RCRA Subtitle D, which mandates engineered composite liners, leachate collection systems, and landfill gas controls (Resource Conservation and Recovery Act, 1976e). C&D landfills, by comparison, are subject only to "minimal" federal criteria, with individual states determining whether more stringent requirements apply (EPA, 2016). This suggests that (a) leachate and gas collection at Swiss landfills may be less rigorous than at modern U.S. MSW landfills, and (b) U.S. landfills receiving construction wood vary widely in the aggressiveness of their leachate and gas management systems, with MSW landfills subject to the most stringent federal oversight. Additionally, U.S. landfills increasingly recirculate leachate to accelerate waste decomposition (EPA, 2023a), though the implications of this practice for eutrophication impacts remain unassessed.

In sum, the conclusions regarding high eutrophication impacts from landfilling MTP in this research depend entirely on the assumption that Swiss landfill technology from 2000 is representative of average modern U.S. landfill conditions and regulatory requirements. This assumption obscures important differences between MSW and C&D landfill practices in the U.S. Future versions of the DATASMART package should incorporate multiple landfill types and provide users with a national average technology mix tailored to specific waste streams.

Cumulative EOL Impacts

The cumulative EOL phase impacts for each building were estimated under a business-as-usual (BAU) waste distribution scenario. The BAU scenario assumed waste distribution data for general C&D wood waste would be applicable to MTP; a more robust assessment would use empirical waste distribution data specific to MTP after a full building lifespan. Until such data become available, this analysis provides an interim approach based on the best available information for C&D wood.

An additional Optimistic waste distribution scenario was considered for MT1, in which structural MTP is recovered for reuse. A reuse rate of 99% was adopted based on the mockup deconstruction results. However, the mockup MTP experienced different environmental conditions than MT1's MTP. In the mockup, MTP was exposed to weathering, which accelerates deterioration, but the mockup was only erected for a few years—much shorter than MT1's 75-year service life.

Despite these differences, other studies suggest that high recovery (95-99%) rates for structural timber are feasible after longer lifespans (Cramer & Ridley-Ellis, 2020; Cramer & Sandin, 2021). These findings are supported by research showing that glulam recovered after a 75-year service life exhibited minimal degradation in structural performance (Rammer et al., 2014). However, these data come from residential buildings, pilot projects, or mechanical testing studies. There is limited quantitative reuse data for MTP in commercial-scale buildings, North American structures, or newly constructed mass timber buildings. Collecting such data as these buildings reach end-of-life would strengthen the assumptions used in the Optimistic scenario.

Conclusions

In the end-of-life (EOL) phase of the product life cycle, the impacts of building deconstruction or demolition (C1), waste transportation (C2), waste processing (C3), and waste disposal (C4) were evaluated for mass timber panels (MTP) and other construction and demolition (C&D) materials used in the MT1 and Concrete Baseline case study buildings. Five EOL scenarios—reuse, reprocess, recycle, incinerate, and landfill—were assessed for MTP, while three scenarios—recycle, incinerate, and landfill—were assessed for other C&D materials. Primary and secondary data were collected and refined for each module to quantify environmental impacts. Material-level C2–C4 impacts were compared across EOL scenarios to evaluate the benefits of diverting waste from landfill, and whole-building C1–C4 impacts were compared under different waste distribution scenarios.

In the first module (C1), demolition was modeled for both buildings using an average of four excavator energy consumption models, while deconstruction was modeled for MT1 using a hand-deconstruction model developed from primary data collected during the mockup deconstruction. Although the hand model is not feasible at full-building scale, it illustrates the upper bound of potential reuse benefits. Dismantling impacts were driven primarily by diesel consumption. Deconstructing MT1 instead of demolishing it reduced impacts across all categories, while demolishing the Concrete Baseline produced the lowest impacts overall. A sensitivity analysis showed that deconstruction impacts are highly dependent on the method used to estimate equipment energy consumption, underscoring the need for primary data on equipment operation in future LCAs.

In the second module (C2), a geospatial model was developed in ArcGIS to estimate transportation distances for MTP and other C&D wood waste across five EOL scenarios. Surveys conducted in five major U.S. cities revealed that the ArcGIS model substantially underestimated transportation distances at the city scale. This suggests that existing hauler relationships, facility capacity, and disposal costs are more influential than geographic proximity in determining where waste is sent.

In the third module (C3), primary LCI data were collected for MTP reuse and reprocessing, including de-nailing, resizing, planing, and sanding. These processes prepare MTP either for structural reuse or for aesthetic applications such as countertops and shelving. The scarcity of primary LCA data on wood reuse—particularly for MTP—limits broader validation and highlights the need for additional data collection from reuse facilities. In contrast, C3 impacts for recycling MTP, based on secondary data, aligned more closely with published LCAs for wood recycling due to the greater availability of data for comparable products.

In the final module (C4), disposal impacts were evaluated using resin-to-wood ratios for U.S.-produced CLT and GLT. Results showed that incineration impacts are sensitive to resin content, which influences fossil CO_2 emissions. However, the analysis relied on polyurethane as a proxy for other resin types, indicating a need for LCI data on combustion of additional resin chemistries. Limited acceptance of engineered wood at U.S. incinerators further constrains disposal options for MTP.

Comparing the cumulative C2–C4 impacts of waste transportation, processing, and disposal per kilogram of material showed that recycling generally produced lower impacts than landfilling for most materials. For MTP, reuse and landfill produced the highest impacts among the five scenarios, while recycling offered a clear environmental advantage. For other materials, recycling outperformed landfilling in nearly all categories, with exceptions for wood and gypsum due to fossil-intensive recycling processes. Transitioning to renewable energy sources could reduce these impacts, as fossil energy consumption drives the majority of recycling burdens.

Whole-building C1–C4 impacts were evaluated under a business-as-usual (BAU) scenario, in which both buildings were demolished and materials were distributed to EOL pathways based on national averages. An additional Optimistic scenario was evaluated for MT1, in which the building was selectively deconstructed to recover structural MTP for reuse. In both buildings, waste processing (C3) and disposal (C4) dominated EOL impacts due to fossil-intensive recycling processes and emissions from landfill decomposition.

Comparing MT1's BAU and Optimistic scenarios showed that selective deconstruction and reuse reduced overall EOL impacts in most categories. AP and SFP increased in the Optimistic scenario due to diesel consumption during reuse processing, indicating an opportunity to reduce impacts by transitioning reuse facilities away from fossil-based energy. The Concrete Baseline exhibited similar or lower impacts than MT1 under both waste scenarios because dismantling concrete structures consumed less diesel than dismantling wood structures. However, this conclusion is limited by the lack of consensus in published literature on whether demolition energy consumption is higher for wood or concrete buildings.

Key Takeaways

This chapter provides a comprehensive assessment of end-of-life (EOL) processes for mass timber products (MTP) and other construction and demolition (C&D) materials, integrating primary data collection, geospatial modeling, and comparative LCA across multiple waste treatment pathways. Several overarching findings emerged:

1. **Across the five MTP EOL scenarios, recycle consistently yields the lowest impacts, while reuse and landfill yield the highest.**
   When C2–C4 impacts of each scenario are compared for MTP, recycle produces the lowest emissions because it avoids both waste disposal burdens and consumes less diesel during waste processing compared to the reuse and reprocess scenarios.
2. **Selective deconstruction reduces whole-building EOL impacts for MT1.**
   In the Optimistic scenario, the building was deconstructed and 99% of structural MTP is recovered for reuse, reducing MT1's total EOL impacts relative to the BAU scenario in most impact categories. Diesel use during reuse processing increased acidification and smog formation impacts compared to BAU, highlighting a potential area of improvement by decreasing reliance on fossil fuels.
3. **Overall, MTP buildings have greater potential for low-impact EOL pathways than conventional concrete buildings.**
   When combined with design-for-disassembly principles and access to reuse markets, MTP enables high-value recovery that is not feasible for reinforced concrete. This advantage becomes especially pronounced when reuse pathways are available within reasonable transportation distances.

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Chapter 3: Additional Environmental Benefits of Production Substitution and Carbon Storage

Introduction

Outside of the cradle-to-grave system boundary, additional environmental benefits and burdens may be accounted for, including production substitution impacts and carbon storage benefits. Product substitution occurs when construction and demolition (C&D) waste is recovered from the landfill and directed toward reuse, recycling, or incineration within the circular economy, thereby displacing the production of virgin materials that would otherwise fulfill the same functional role. Beyond substitution, circular economy pathways can provide broader environmental and social advantages—such as improved resource efficiency, reduced extraction burdens, extended carbon storage, and local job creation—that traditional LCA does not capture.

The type of virgin product displaced by C&D waste retained in the circular economy depends on both the waste material and the selected end-of-life (EOL) pathway. For example, reusing or reprocessing wood products such as mass timber panels (MTP) reduces the need for harvesting new fibers and manufacturing new MTP. Recycling wood waste similarly displaces the extraction of virgin wood residues; however, because recycling replaces only the feedstock and not the finished product, the impacts of manufacturing a new panel are not displaced. Incinerating wood waste with energy recovery provides a different form of substitution by displacing fossil fuel combustion for electricity.

In life cycle assessment (LCA) studies, ISO 21930:2017 allows reporting of benefits beyond the system boundary under Module D, including avoided impacts from displacing virgin materials or energy production (ISO, 2017). MTP LCA studies that include Module D vary in their conclusions about the most environmentally preferable EOL scenario: some identify reuse as yielding the lowest embodied carbon (Greene et al., 2023; Passarelli, 2018), while others find incineration preferable depending on the displaced energy source (Morris, 2017). These differing outcomes reflect the sensitivity of Module D results to assumptions about the types and quantities of products displaced by recovered MTP.

Studies examining other C&D waste materials show similar dependence on substitution assumptions. Some analyses that include avoided primary production of virgin aggregate, metals, or glass find that substitution benefits generally do not outweigh the additional burdens of waste processing across most impact categories, with the exception of eutrophication (Borghi et al., 2018; Ruggeri et al., 2019). At the whole-building scale, recycling benefits were insufficient to offset the increased equipment use associated with deconstruction relative to demolition, resulting in higher net impacts for deconstruction (Ruggeri et al., 2019). Conversely, other studies have shown that including Module D in a cradle-to-grave scope can offset up to 50% of a building's environmental impacts, particularly eutrophication (Delem & Wastiels, 2019). Comparisons between MTP and mineral construction further indicate that MTP can achieve larger Module D reductions in global warming impacts, though smaller reductions in other categories (Delem & Wastiels, 2019). Comparable analyses have not yet been conducted using North American waste management practices and energy profiles.

Biogenic carbon storage is another benefit commonly included in MTP LCA studies, though this is not shared by conventional mineral construction materials. Wood stores carbon absorbed during tree growth, with approximately 50% of its mass composed of carbon (Bergman et al., 2014). At EOL, this carbon may be released (incineration, landfill), transferred to another product system (reuse, reprocess, recycling), or permanently stored (landfill). ISO 21930:2017 and ISO 13391:2025 account for biogenic carbon flows to and from the atmosphere under the assumption that all biogenic carbon released back to the atmosphere after being sequestered by biomass will be recaptured by regrowing biomass, provided that annual harvest levels do not exceed growth (ISO, 2017; ISO, 2025). As a result, biogenic CO_2 emissions are excluded from the impact assessment stage in LCA.

Several approaches have been developed to address temporary biogenic carbon storage over varying time periods, rather than only assessing permanent storage benefits like ISO standards do. Postponing biogenic CO_2 emissions by temporarily storing them in wood products can mitigate climate change by postponing radiative forcing and keeping atmospheric CO_2 concentrations low, which has short-term benefits of “buying time” for climate change mitigation technologies and strategies to develop (Levasseur et al., 2012).

Ton-year systems such as the Moura-Costa and Lashof methods assign credits to carbon stored in a system for a given duration, using equivalency factors to convert ton-years of storage into an amount of CO_2 emissions avoided (Levasseur et al., 2012). Under both approaches, equivalency factors are calculated by integrating the cumulative radiative forcing of a 1-ton CO_2 emission over a 100-year time horizon.

The Moura-Costa approach applies an equivalency factor of 48 years, meaning that removing and storing 1 ton of CO_2 for 48 years is considered equivalent to avoiding a 1-ton CO_2 pulse emission over 100 years (Moura-Costa & Wilson, 2000). Storage credits are distributed evenly across each year of storage (0.02 tons CO_2 per year). In contrast, the Lashof approach assumes that storing carbon provides the same benefit as delaying its emission until the end of the storage period (Fearnside et al., 2000). Storage benefits are calculated as the difference between the integrals of the emissions curves for an immediate emission and an emission delayed to the actual year of release. For example, storing carbon until year 48 yields a benefit equal to the area under the curve from year 100 to year 148 (Brandão et al., 2013). These benefits are not distributed uniformly over the storage period and depend on the length of the delay (Levasseur et al., 2012).

Both the Moura-Costa and Lashof approaches have limitations in their treatment of emission timing, as temporal effects are muted when emissions are aggregated in the life cycle inventory (Levasseur et al., 2012). Levasseur et al. (2010) addressed this limitation with dynamic LCA, which uses dynamic inventories and time-dependent characterization factors to account for the timing of emissions over flexible time horizons. Using dynamic LCA instead of ton-year approaches enables consideration of all life cycle stages, which is particularly relevant in this analysis because different EOL treatments for MTP determine how much biomass remains stored versus emitted.

An additional environmental benefit associated with construction materials is concrete carbonation. Cement-containing materials (CCM) in concrete can re-sequester a portion of the CO_2 released during cement production through carbonation, a chemical reaction in which CO_2 reacts with calcium oxide (CaO) to form calcium carbonate ($CaCO_3$) (AzariJafari et al., 2021). Unlike wood, which does not sequester additional carbon after harvesting, concrete continues to carbonate throughout its use and EOL phases. The extent of carbonation depends on environmental conditions (e.g., moisture, temperature) and physical properties of the concrete mix (e.g., binder type, curing time, water content) (AzariJafari et al., 2022).

A key distinction between wood carbon storage and concrete carbonation lies in the timing and type of carbon flows. Wood stores biogenic carbon during tree growth and releases it at EOL. Concrete releases fossil CO_2 during cement production (calcination) and later reabsorbs a portion of that CO_2 through carbonation. Calcination and carbonation are therefore inverse processes. Due to the higher storage potential of wood, an MTP building can store roughly five times more carbon through the EOL phase than its alternate concrete design (Liang et al., 2021).

Research Objectives

Addressing the data gaps identified in existing Module D and carbon storage LCA research is essential for improving understanding of MTP's environmental impacts across its full life cycle and its comparative performance relative to other C&D materials. This chapter evaluates the Module D environmental impacts associated with a case-study MTP building ("MT1") and an alternative concrete design ("Concrete Baseline"). This is achieved by compiling existing data, validating assumptions through literature and surveys, and comparing environmental impacts by material type. An additional analysis highlights differences among existing carbon storage accounting methods and examines how each approach influences the comparative environmental performance of different EOL scenarios for MTP.

The research objectives are to:

1. **Evaluate Module D benefits** for all applicable materials in reuse, reprocess, recycle, and incinerate scenarios.
2. **Compare biogenic carbon storage benefits for MTP** across EOL scenarios using two carbon storage accounting methods.
3. **Evaluate carbonation benefits** for reinforced and unreinforced concrete throughout the cradle-to-grave lifespan.

The overarching goal of this chapter is to assess the environmental benefits and burdens associated with recovering C&D waste from the landfill, and to compare different carbon storage accounting methods for wood waste. Together, these analyses improve understanding of the environmental tradeoffs and benefits associated with each material under different EOL fates. This research provides foundational data that can strengthen future LCA studies incorporating Module D benefits for recoverable C&D materials and offers insight into gaps in current carbon storage accounting guidelines.

Methods

Goal & Scope

This LCA was conducted in accordance with ISO 14040, ISO 14044, and ISO 21930 (ISO, 2006a-b; ISO, 2017). All impacts were modeled in SimaPro v9.5 using the TRACI characterization method (Bare, 2012). The functional unit for this analysis is one building.

The environmental burdens and benefits associated with displacing virgin products using reclaimed construction and demolition (C&D) waste are quantified in Module D. Figure 33 illustrates the system boundary and shows how processes and materials are included in each life cycle phase. While the virgin product substitution impacts are only evaluated in Module D, carbon storage benefits are evaluated starting in the production phase (A1) and ending after EOL (C3-C4).

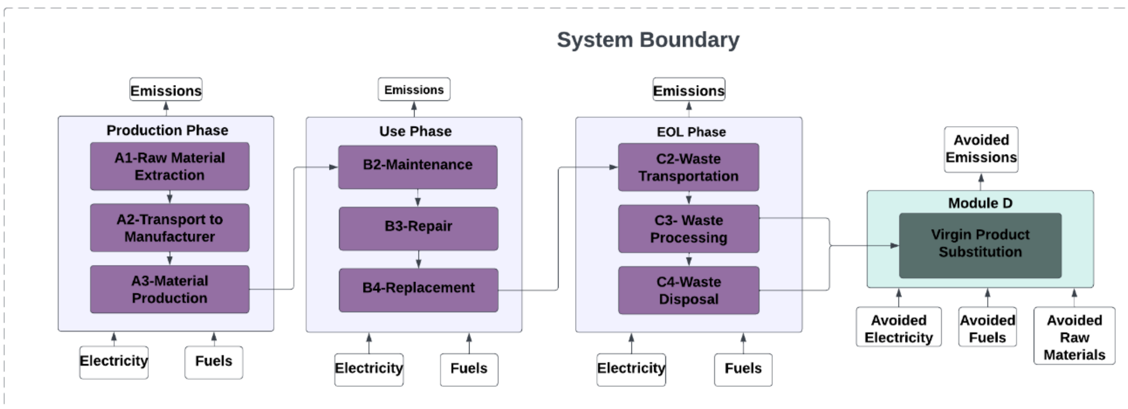

Figure 33. System boundary diagram for carbon storage and virgin product substitution by recovered construction and demolition waste.

This study evaluates the five mandatory impact categories specified in ISO 21930:2017 (Table 32). Two additional inventory parameters (biogenic carbon removals and emissions from products) were included to track biogenic carbon flows across the system boundary.

Table 32. Impact category indicators per ISO 21930 (ISO, 2017).

Impact Categories	Abbreviation	Units	Method
Global warming potential, fossil	GWP_{FOSSIL}	kg CO_2e	TRACI 2.1 V1.08
Acidification potential	AP	kg SO_2e	TRACI 2.1 V1.08
Eutrophication potential	EP	kg Ne	TRACI 2.1 V1.08
Ozone depletion potential	ODP	kg CFC-$_{11}$e	TRACI 2.1 V1.08
Smog formation potential	SFP	kg O_3e	TRACI 2.1 V1.08
Global warming potential, biogenic	$GWP_{BIOGENIC}$	kg CO_2e	TRACI 2.1 V1.08 + LCI Indicatory
Biogenic carbon removal from product	BCRP	kg CO_2e	LCI Indicatory
Biogenic carbon emission from product	BCEP	kg CO_2e	LCI Indicatory

Because recycled materials are included in the analysis, the avoided burden (0:100) allocation approach was adopted (De Wolf et al., 2020). Under this method, all burdens and benefits associated with recycling are assigned to the first life cycle of the material, rather than the secondary product. This enables explicit calculation of Module D impacts and aligns with common practice for closed-loop recycling systems such as steel and aluminum, where scrap is returned to primary production to create the same product type (World Steel Association, 2017).

Displacement of Virgin Materials (Module D)

Module D impacts represent the avoided burdens associated with manufacturing virgin products (A1–A3) that are displaced by recovered materials. Under ISO 21930:2017, Module D is calculated as the difference between the output flows of recovered materials and the input flows of the displaced virgin materials (ISO, 2017). For recoverable C&D materials in MT1, Module D impacts were determined by identifying the virgin products displaced by reused, reprocessed, recycled, or incinerated C&D waste, and subtracting the A1–A3 impacts of producing those avoided products.

This analysis was conducted for MTP and other non-MTP wood products, steel, rebar, aluminum, concrete, and gypsum, as these materials are reported to be recovered from the landfill at a U.S. average level (EPA, 2020). Only MTP and non-MTP wood were considered eligible for reuse, reprocessing, recycling, or incineration; all other materials were assumed to be recycled only. The methodology for calculating Module D impacts for each recoverable material is described in the following subsections, beginning with MTP.

MTP & Wood

In Chapter 2 of this dissertation, five EOL scenarios were evaluated for MTP: reprocess, reuse, recycle, incinerate, and landfill, with all but landfill representing circular economy pathways. Module D impacts for these circular scenarios are calculated separately because each one displaces a different type of virgin product.

Non-MTP wood products such as lumber and plywood can also be recycled or incinerated. Their Module D impacts were assumed to be identical to those of MTP because recycled wood products are converted into residues with similar next uses, and because the energy content of wood is consistent across product types despite differences in resin content.

The methodology for calculating Module D impacts for MTP (and, where applicable, other wood products) under each EOL scenario is described below, beginning with reprocessing.

Reprocess

In the reprocess scenario, salvaged MTP displaces wood products used in aesthetic applications such as benches, countertops, or exposed non-structural beams. In Chapter 2, different reuse centers were surveyed about their reuse and reprocess practices for salvaged MTP. One facility, Earthwise, reported that butcher block countertops are the primary use for their salvaged GLT, and because Earthwise provided the primary data for this scenario, butcher block was selected as the displaced product.

Published LCA studies for butcher block countertops are not available, so several assumptions were required. Butcher block manufacturing processes were reviewed using information from John Boos & Co., which produces both edge-grain and blended-grain butcher block in Illinois (“How

Butcher Block Is Made," n.d.). Their process includes kiln drying and planing lumber, sawing boards into 1.75-inch rails, finger-jointing and laminating the rails in a screw press, and sanding the final product. Blended-grain blocks use rails of varying lengths to create a checkerboard appearance, while edge-grain blocks use uniform rails (Figure 34).

This manufacturing process is similar to that of virgin GLT, which uses green lamstock, dries and finger-joints boards, face-bonds them using pressure and radio-frequency or cold curing, and planes the resulting beams. The primary difference lies in product layup rather than processing steps. Because of these similarities, the A1–A3 impacts of producing 1 kg of virgin GLT were used as a proxy for the impacts of producing 1 kg of butcher block. LCA data for virgin GLT production was taken from Puettmann (2025a).

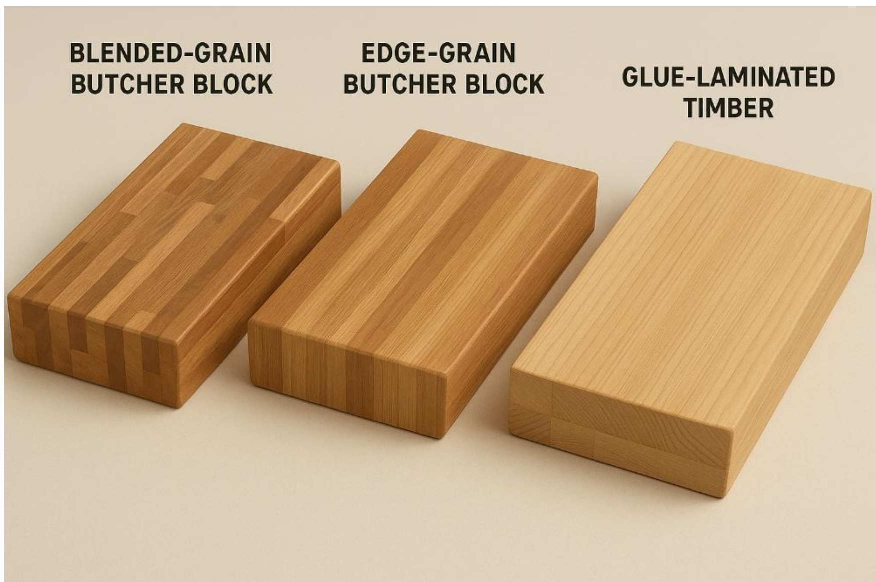

Figure 34. Side-by-side comparison of two types of butcher block and glued-laminated timber (GLT). Generated by ChatGPT-5.

Reuse

In the reuse scenario, salvaged GLT displaces wood products used in both aesthetic and building applications. For simplicity, it is assumed that un-planed salvaged GLT is reused in buildings as either structural or non-structural elements. Therefore, the displaced product is virgin GLT. Module D impacts were calculated using LCA data for virgin GLT production from Puettmann (2025a).

Recycle

Recycled MTP and other non-MTP wood products displace feedstock used in composite wood products such as particleboard. U.S. particleboard is produced from both virgin and recycled residues (Puettmann, 2024a). Approximately 95 percent of residues originate from virgin sources (whole logs, chips, sawdust, shavings, plywood trim), and 5 percent from recycled sources (construction waste, recycled particleboard, unspecified). Of the virgin residues, 70 percent are green (generated before drying) and 30 percent are dry (generated after drying). Regional contributions include 15 percent from the Pacific Northwest (PNW), 35 percent from the Southeast

(SE), 43 percent from the Northeast–Northcentral (NE-NC), and 7 percent from the Inland Northwest (INW).

For Module D, recycled MTP or non-MTP wood is assumed to displace virgin residues from lumber and plywood facilities. A mass-balance approach was used to determine the quantity of each residue type displaced by 1 kg of recycled MTP. Puettmann (2024a) reports oven-dry residue quantities used to manufacture 1 m^3 of particleboard. These data were used to calculate the distribution of residues by moisture content and region (

Table 33), and to determine the oven-dry mass of each residue type displaced per kilogram of recycled MTP.

Table 33. Distribution of virgin wood residue types displaced by using 1 oven-dry kilogram (odkg) of recycled mass timber panel (MTP) residues, by moisture content and by region in the Pacific Northwest (PNW), Inland Northwest (INW), Southeast (SE) and Northeast-Northcentral (NE-NC) regions.

Region	Moisture Content	Products	Percent Distribution	Quantity Displaced per kg Recycled MTP (odkg)	LCI Data Source
PNW	Green	Log, chips, sawdust, shavings	10%	0.10	Puettmann, 2024b
PNW	Dry	Chips, sawdust, shavings, plywood trim	5%	0.05	Puettmann, 2024b
INW	Green	Log, chips, sawdust, shavings	5%	0.05	Puettmann, 2024c
INW	Dry	Chips, sawdust, shavings	2%	0.02	Puettmann, 2024c
SE	Green	Log, chips, sawdust, shavings	24%	0.24	Puettmann, 2024d
SE	Dry	Chips, sawdust, shavings, plywood trim	11%	0.11	Puettmann, 2024d
NE-NC	Green	Log, chips, sawdust, shavings	30%	0.30	Puettmann, 2025b
NE-NC	Dry	Chips, sawdust, shavings, plywood trim	13%	0.13	Puettmann, 2025b
Total			*100%*	*1.00*	

Environmental impacts for each displaced residue type were calculated using updated LCI data from Puettmann (2024b-d, 2025b). The LCI for sawing rough green lumber (producing green residues) and planing dry lumber (producing dry residues) was modeled in SimaPro using mass allocation between products and co-products. Impacts of harvesting softwood sawlogs were modeled using regional forestry LCI data. Impacts of plywood trim were modeled using regional plywood trimming processes in the DATASMART database (LTS, 2023). Impacts per kilogram of each residue type were multiplied by the displaced quantities in

Table 33 and summed to obtain the Module D impacts for the recycle scenario.

Incinerate

Although incineration does not retain MTP or non-MTP wood products in the circular economy, it can provide an avoided-burden benefit by displacing fossil-based electricity generation when waste-to-energy facilities recover energy from combustion. Module D for incineration therefore differs from the other scenarios.

The EPA WARM tool calculates Module D for incineration using an avoided utility emission offset (EPA, 2023b). The calculation incorporates:

- The energy content of wood (5.36 kWh/kg)
- The combustion system's energy conversion efficiency (17.8 percent)
- The avoided emissions from generating an equivalent amount of electricity from the U.S. grid

Avoided emissions were modeled using an electricity process from the Ecoinvent database based on the average U.S. grid mix (Wernet et al., 2014). Multiplying these factors yields the Module D impacts for the incinerate scenario.

Steel/Rebar

The World Steel Association provides guidance for calculating substitution impacts for recycled steel using the avoided burden cut-off approach (World Steel Association, 2017). Steel can be produced via two routes:

- Primary production in a blast oxygen furnace (BOF), which relies mainly on virgin iron but may include up to 30% scrap.
- Secondary production in an electric arc furnace (EAF), which uses scrap as the primary feedstock with small additions of virgin iron depending on the steel grade.

Because U.S. steel production is dominated by the EAF route, the World Steel Association concludes that recycled steel displaces primary steel produced via the BOF route. Thus, the displaced product for recycled steel is primary crude steel.

The background LCA report for the U.S. industry-average rebar EPD (representing 70% of U.S. production) describes the EAF-based manufacturing process, which includes melting scrap and other feedstock materials in the EAF, refining in a ladle metallurgy furnace, casting into blooms or billets, and rolling into rebar (AISC, 2022). Since crude steel slabs are the first "recognizable" product in the system and can be further processed into rebar or other steel products, the World Steel Association recommends calculating substitution impacts at the crude steel stage.

Module D impacts for recycled steel were therefore calculated by subtracting the impacts of producing secondary crude steel from the impacts of producing primary crude steel. These were estimated using the DATASMART processes "Steel, electric, un- and low-alloyed, at plant" and "Steel, converted, unalloyed, at plant," respectively (LTS, 2023).

Aluminum

Aluminum recycling requires a separate substitution analysis because aluminum is a non-ferrous metal and follows different production and recycling pathways than steel. Aluminum manufacturing produces an intermediate product—aluminum ingots—that serve as precursors to construction products such as extrusions and sheet aluminum (The Aluminum Association, 2022a). Primary and secondary production routes differ substantially in both material inputs and technology.

In the primary route, alumina (Al_2O_3) is extracted from bauxite ore via the Bayer process, which includes crushing, digestion with sodium hydroxide, clarification, precipitation, and calcination (Walker, n.d.). The resulting alumina is then reduced to aluminum metal through the Hall–Héroult electrolytic process, in which electric currents separate aluminum and oxygen ions; oxygen reacts with carbon anodes to form CO_2, while molten aluminum collects at the cathode.

In the secondary route, pre-sorted aluminum scrap is melted in a furnace, purified with injected salts or gases, alloyed as needed, and cast into ingots (The Aluminum Association, 2022b).

Module D impacts for aluminum were calculated by subtracting the A1–A3 impacts of secondary aluminum ingots from those of primary aluminum ingots. U.S. industry-average EPDs for primary and secondary aluminum ingots were used as the LCI sources (The Aluminum Association, 2022b–c).

Concrete

Module D impacts for recycled concrete were estimated by identifying common end uses for recycled aggregate and determining the type of virgin materials displaced. Concrete consists of cement, aggregate, and water (Hottle et al., 2022). Recycled aggregate is produced by crushing recovered concrete and separating the resulting material into particle size fractions (Wilburn & Goonan, 1998; Zhang et al., 2023). After crushing, contaminants are removed using vibrating feeders and magnetic separators, and the aggregate is screened into coarse (12–22 mm), intermediate (4–12 mm), and fine (<4 mm) fractions.

Recycled aggregate is primarily used as road base (68%) (EPA, 2023b). Therefore, virgin aggregate used in road pavement production was selected as the displaced material. Road pavement consists of multiple layers—sub-base, base, and surface concrete—typically composed of asphalt concrete (Gouveia et al., 2022). In the absence of asphalt concrete LCI data, aggregate consumption statistics for construction, concrete, and asphalt in 2022 were used to estimate the average composition of virgin aggregate: 29% gravel and sand[3], and 71% crushed stone (National Minerals Information Center, 2025).

Based on assumptions from North American studies of recycled aggregate substitution (McIntyre et al., 2009), it was assumed that 1 kg of recycled aggregate displaces 1 kg of virgin aggregate. The impacts of producing the displaced virgin aggregate were estimated using DATASMART processes for each aggregate type and summed to determine Module D impacts (LTS, 2023).

Gypsum

Recycled gypsum is commonly sold as an agricultural soil amendment (EPA, 2023b). Virgin gypsum powder—also referred to as crude gypsum, gypsum mineral, or calcium sulfate dihydrate

[3] Consistent with findings that recycled aggregate substitution reduces concrete compressive strength when replacing sand, only gravel was assumed to be displaced.

($CaSO_4 \cdot 2H_2O$)—is similarly sold directly to agricultural markets (USGS, 2025). Therefore, recycled gypsum is assumed to displace virgin gypsum powder.

Module D impacts for recycled gypsum were calculated by negating the A1–A3 impacts of producing virgin gypsum powder. These impacts were modeled using the DATASMART process for gypsum powder production, which includes mining, crushing, and milling of natural gypsum (LTS, 2023).

Biogenic Carbon

Biogenic carbon storage in MTP is evaluated under two accounting approaches for each waste scenario: the ISO method and the Radiative Forcing discounting method. The ISO method applies the +1/–1 accounting framework specified in ISO 21930 and ISO 13391, resulting in a storage benefit of –1.65 kg CO_2e per kilogram of MTP or wood in the landfill scenario and no storage benefits for other EOL pathways (ISO, 2017; ISO, 2025). This method is also applied to all non-MTP wood products.

The Radiative Forcing approach evaluates the climate benefit of biogenic carbon storage using dynamic radiative forcing methodology. This method discounts the total carbon storage benefit over the product's lifespan, prorated by the duration of storage, thereby avoiding double counting. Because the purpose of this analysis is to illustrate how different methods treat each EOL scenario, this approach is applied only to MTP.

The methodology behind each approach is described in the following subsections, beginning with the ISO method.

ISO Accounting Method

Under ISO 21930, biogenic carbon flows entering the system boundary (removals) are expressed as −1 kg CO_2e per kg of biogenic CO_2, and biogenic carbon leaving the system boundary (emissions) is expressed as +1 kg CO_2e per kg of biogenic CO_2. These flows are reported under the $GWP_{BIOGENIC}$ indicator. Sequestration in the product (BCRP) is reported in C1–Building Deconstruction, while emissions from the product (BCEP) are reported in C3–Waste Processing or C4–Waste Disposal, depending on the EOL scenario. For reuse, reprocess, and recycle, emissions from the product are reported in C3, while emissions from disposed coproducts generated during waste processing are reported in C4. For incinerate and landfill, all emissions are reported in C4.

For reused and reprocessed MTP, a reuse facility (Earthwise) was surveyed in Chapter 2 to determine input and output flows during C3–Waste Processing. Earthwise reported mass losses of 25 percent and 16.67 percent, respectively; these losses are treated as coproduct emissions in C4–Waste Disposal. In the recycle scenario, mass losses may occur after chipping during quality-control checks to ensure that chips meet specifications for moisture content, particle size distribution, and contamination (EPF, 2018). Because no data exist for mass losses in C&D waste processing facilities, national market data for roundwood residues were used as a proxy. The Timber Products Output (TPO) tool reports that, on average, 36 percent of residues are used in new panel manufacturing, 49 percent for fuel, and 14 percent for mulch (USFS, 2025). Since the recycle scenario assumes that chipped MTP is used for new panel manufacturing, a 64 percent mass loss was applied to the biogenic carbon stored in recycled MTP, and this loss was reported as coproduct emissions in C4.

For incineration, 100 percent of the biomass is combusted and emitted, leaving no biogenic carbon in the original system boundary. For landfill, ISO 13391 Section 5.5.2 requires landfill carbon storage to be calculated using Equation 6, assuming a degradable organic carbon fraction of 50 percent (wDOC = 0.5) and a landfill gas conversion factor of 10 percent (wDOC,diss = 0.1) (ISO, 2025). This results in 1.65 kg CO_2e of biogenic carbon remaining in the product per kg of MTP.

Equation 6. Biogenic carbon remaining in landfilled wood.

$$w_{i,rem} = m_{i,ent} * w_{doc} * \left(1 - w_{DOC,diss}\right) * \frac{44}{12}$$

Where:

- $W_{i,rem}$ = biogenic carbon in landfilled wood materials (MT CO_2e)
- $m_{i,ent}$ = mass of wood entering landfill (MT)
- W_{doc} = degradable organic carbon fraction
- $W_{doc,\ diss}$ = fraction converted to landfill gas
- $\frac{44}{12}$ = carbon-to-CO_2 conversion factor

Across a 100-year time horizon, biogenic carbon stored in reused, reprocessed, or recycled MTP is carried over the system boundary into the next product system. The "first life" of MTP in MT1 is assumed to be 75 years, and the "second life" of recovered MTP is accounted for in the remaining 25 years. Under ISO 21930, the biogenic carbon flows in the next product system are expressed as −1 kg CO_2e per kg of biogenic CO_2. It is assumed that the next products have functional lifespans longer than 25 years, so emissions from the system boundary are not included. This creates a net negative carbon storage result for recovered MTP.

Radiative Forcing Accounting Method

In this accounting method, a dynamic radiative forcing (RF) approach is adopted to assess biogenic carbon storage benefits for MTP under each EOL treatment. The RF of avoided emissions is calculated annually by multiplying the radiative efficiency (RE) of CO_2 by the decay function for a 1 kg pulse emission of CO_2 (Equation 7). Summing the annual RF values yields the net cumulative radiative forcing (NCRF) (Equation 8). Dividing the NCRF of stored carbon by the NCRF of an equivalent emission yields the avoided GWP (Equation 9), which is negative because the carbon has not been emitted.

Equation 7. Radiative forcing of stored biogenic CO_2 at time *t*.

$$RF_{{CO_2}^{bioCS}}(t) = -RE_{CO_2} * [CO_2](t)$$

Equation 8. Net cumulative radiative forcing for *m* years of storage.

$$NCRF^{0\ to\ m}_{{CO_2}^{bioCS}} = \sum_{0}^{m} RF_{{CO_2}^{bioCS}}(t)$$

Equation 9. Avoided GWP for *m* years of storage.

$$GWP_{BIOGENIC_t} = NCRF^{0\ to\ m}_{CO_2\ bioCS} \,/\, \sum_{0}^{t} NCRF_{CO_2}(t)$$

A 100-year time horizon was selected to illustrate temporary storage benefits across multiple product systems. It was assumed that MTP leaves the first product system after 75 years, consistent with MT1's service lifespan. RF values for the first 75 years were calculated using the initial biogenic carbon content of MTP (−1.83 kg CO_2e/kg). RF values for the remaining 25 years were calculated based on the amount of biogenic carbon remaining after waste processing and new product manufacturing in each EOL scenario. These values are shown in Table 34.

Table 34. Biogenic carbon stored in years 75–100 for five end-of-life (EOL) scenarios for mass timber panels (MTP) in a case study building.

EOL Scenario	Biogenic Carbon Stored in Years 75-100 (kg CO_2/kg MTP)
Reuse	1.37
Reprocess	1.53
Recycle	0.52
Incinerate	0.00
Landfill	1.65

The values in Table 34 were derived from various assumptions about mass losses of MTP after going through waste processing and/or disposal in each EOL scenario. In the reuse and reprocess scenarios, the previously reported mass losses of 25 percent and 16.67 percent were applied to Equation 7. In the recycle scenario, MTP residues are assumed to be used in new panel products such as particleboard. In addition to the 64 percent mass loss during waste processing, a 20 percent mass loss during new panel manufacturing was applied, based on losses reported for particleboard produced from recycled residues (Kim & Song, 2014).

In the incinerate scenario, all stored biogenic CO_2 is emitted in year 75. In the landfill scenario, 10 percent of stored biogenic CO_2 is emitted during methanogenic decay, per ISO 13391 (Equation 6) (ISO, 2025). As a conservative assumption, this biogenic carbon is assumed to be lost immediately, even though landfill decay is slow and depends on environmental conditions and landfill management practices (EPA, 2023a). While this assumption does not reflect actual decay dynamics, it provides a conservative estimate of the storage benefit and reduces uncertainty associated with variable landfill conditions.

Concrete Carbonation

Sacchi and Bauer (2020) describe calcination as the decomposition of limestone ($CaCO_3$) into CaO and CO_2 during clinker production (Equation 10). After calcination, hydration occurs when water is added to cement, forming calcium hydroxide ($Ca(OH)_2$) (Equation 11). Carbonation then occurs as $Ca(OH)_2$ reacts with atmospheric CO_2 to form $CaCO_3$ (Equation 12).

Equation 10. Calcination of limestone into calcium oxide.

$$CaCO_3 + Heat \rightarrow CaO + CO_2$$

Equation 11. Hydration of calcium oxide into calcium hydroxide.

$$CaO + H_2O \rightarrow Ca(OH)_2$$

Equation 12. Carbonation of calcium hydroxide into calcium carbonate.

$$Ca(OH)_2 + CO_2 \rightarrow CaCO_3 + H_2O$$

The amount of CO_2 released during calcination is not fully re-sequestered during carbonation. Hydration yields may be reduced by CaO lump formation or premature precipitation into limestone (Sacchi & Bauer, 2020). A global study by Xi et al. (2016) estimates that only 43% of calcination-related CO_2 emissions are reabsorbed through carbonation. Lagerblad (2005) further recommends assuming a 75% hydration yield, meaning only 75% of cementitious particles are available to carbonate throughout the use and EOL phases.

EN 16757 provides an equation for estimating maximum CO_2 uptake during carbonation (Equation 13), based on material properties, exposure conditions, and time.

Equation 13. Maximum carbonation CO_2 sequestration of cement-containing materials, for a given surface *i*.

$$\mathrm{CO_2\ uptake} = \sum (k_i * DOC_i * A_i) * \frac{\sqrt{t}}{1000} * U_{tcc} * C,$$

Where:

- k_i = carbonation rate factor,
- DOC_i = degree of carbonation,
- A_i = surface area,
- t = time (yr),
- U_{tcc} = maximum theoretical kg CO_2 per kg cement,
- C = cement content per m^3 of concrete

To compare concrete's carbonation benefit with MTP's carbon storage on an equivalent basis, CO_2 uptake during both the use phase and EOL must be quantified. For the use phase, Equation 13 was applied to 1 kg of ready-mix concrete, assuming a cement content of 10–15%. Table 35 lists the variables used.

Table 35. Variables used to calculate carbon dioxide uptake per cubic meter (m^3) of ready-mix concrete.

Variable	Value & Unit	Source
k_i	2.7 mm/√(t)	EN 16757:2022
DOC_i	40%	EN 16757:2022
A_i	60 m^2	Sacchi & Bauer, 2022
t	75 yr	Assumed
U_{tcc}	0.49 kg CO_2/kg cement	EN 16757:2022
C	300 kg cement/m^3 concrete	Assumed

During the EOL phase, carbonation increases because crushing exposes uncarbonated surfaces and raises surface-to-volume ratios. Particle size distributions, storage conditions, and exposure durations vary widely across waste processors, introducing uncertainty into EOL estimates. Stripple and Gustafsson (2018) estimate 10 kg CO_2/m^3 (0.0042 kg CO_2/kg) of carbonation during waste processing. After processing, concrete is either landfilled (C4) or recycled as road base (Module D), with an additional 10 kg CO_2/m^3 (0.0042 kg CO_2/kg) of carbonation occurring during secondary use. Landfilled concrete is assumed not to carbonate further because buried particles lack air exposure.

Whole Building Benefits

The Module D impacts and carbon storage benefits associated with each case study building were assessed under the waste scenarios developed in Chapter 2:

- A business-as-usual (BAU) scenario, in which both buildings were demolished and C&D materials were distributed to EOL pathways based on national averages.
- An Optimistic scenario for MT1 only, in which the building was selectively deconstructed to recover structural MTP for reuse.

Module D impacts were calculated for all non-landfilled materials in each building by applying the Module D emission factors for MTP, non-MTP wood products, aluminum, concrete, gypsum, and steel. The cumulative Module D impacts depend on the waste distribution scenario applied. For each material, the share of mass allocated to each applicable EOL scenario (e.g., recycle, reuse, reprocess, incinerate) is multiplied by the Module D impact factor for that scenario and then by the total mass of the material in each building.

The cumulative carbon storage benefits under each accounting method were also calculated for each building. For MT1, the ISO and Radiative Forcing carbon storage benefits are calculated separately for the BAU and Optimistic scenarios, reflecting differences in the distribution of MTP to the five EOL pathways. For the Concrete Baseline, only the ISO method is applied, since the Radiative Forcing approach is used solely to evaluate the influence of increased MTP reuse in MT1.

Lastly, concrete carbonation is modeled using a single approach across all scenarios, based on the cradle-to-grave sequestration per kilogram of concrete evaluated with Equation 13. Because this value is identical for recycled and landfilled concrete, carbonation results do not vary between BAU and Optimistic scenarios for each building.

Results & Discussion

The LCIs for each module of the building life cycle were modeled in SimaPro v9.5 using processes from the DATASMART LCI database (LTS, 2023). Five impact categories were evaluated using the TRACI characterization method (Bare, 2012): fossil global warming potential (GWP_{FOSSIL}), ozone depletion potential (ODP), acidification potential (AP), eutrophication potential (EP), and smog formation potential (SFP). An additional impact category (biogenic global warming potential, $GWP_{BIOGENIC}$) was evaluated manually under the ISO and the Radiative Forcing carbon storage accounting methods. The LCIA results for Module D, biogenic carbon storage, and concrete carbonation are presented separately, starting with Module D.

Module D

MTP & Wood

The Module D LCIA results for MTP in the circular economy EOL scenarios—reprocess, reuse, recycle, and incinerate—are presented in Table 36. The recycle and incinerate results also apply to other non-MTP wood products evaluated in this study, such as lumber and plywood.

The reprocess and reuse scenarios have the largest Module D impacts in the acidification (AP), ozone depletion (ODP), and smog formation (SFP) categories, indicating that avoiding virgin MTP production yields the greatest benefits for these impacts. These categories are discussed below to clarify why displacement of virgin MTP results in comparatively large avoided burdens.

Table 36. Module D impacts for treating 1 kilogram (kg) of waste mass timber panels (MTP) and other wood products in different end-of-life (EOL) scenarios, calculated as the avoided impacts of producing displaced virgin products in each scenario.

EOL Scenario	Product	GWP_{FOSSIL} (kg CO_2e/kg)	AP (kg SO_2e/kg)	EP (kg Ne/kg)	ODP (kg CFC-$_{11}$e/kg)	SFP (kg O_3e/kg)
Reprocess/Reuse	MTP	-2.91E-01	-1.83E-03	-6.58E-04	-2.47E-08	-5.73E-02
Recycle	MTP/ Wood	-7.84E-02	-6.90E-04	-1.75E-04	-7.80E-10	-1.97E-02
Incinerate	MTP/ Wood	-4.56E-01	-1.28E-03	-2.27E-03	-4.84E-09	-1.33E-02

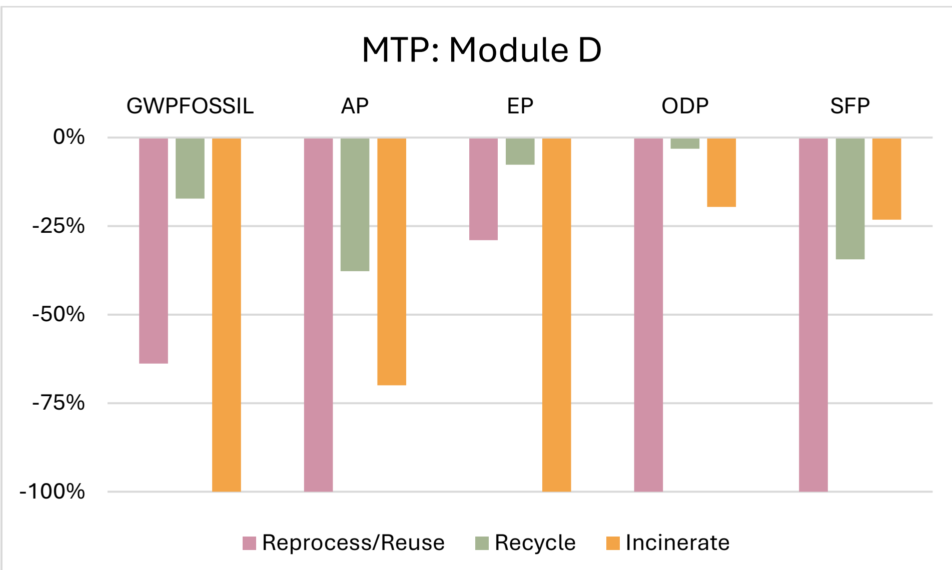

Figure 35. Relative Module D environmental impacts for 1 kilogram of mass timber panel (MTP) waste in different end-of-life (EOL) scenarios, when production of virgin products is avoided. The results are normalized against the EOL scenario with the largest impact in each impact category.

The GWP_{FOSSIL} category reflects contributions of greenhouse gases (GHGs) such as CO_2, CH_4, and nitrogen dioxide (NO_2) to global warming ("Global Warming Potential," n.d.). These gases differ in atmospheric abundance, radiative efficiency, and residence time. When MTP is incinerated for energy recovery, it is assumed that an equivalent amount of electricity from the average U.S. grid is avoided. In 2024, the U.S. grid was powered by 43% natural gas, 19% nuclear, 16% coal, and the remainder renewables ("Electricity generation," n.d.). Avoiding fossil-powered electricity therefore drives the large Module D GWP_{FOSSIL} benefit for the incinerate scenario.

The AP impact category reflects contributions of gases and chemical compounds to acid rain formation, which lowers the pH of soil and water ("Acidification Potential," n.d.). Major contributors include sulfur dioxide (SO_2), dinitrogen monoxide (N_2O), nitrogen monoxide (NO), and other nitrogen oxides (NO_x). During virgin GLT manufacturing, forestry operations emit SO_2 and NO_x due to diesel-powered equipment such as feller bunchers, wheeled skidders, and slide-boom delimbers (Puettmann, 2025a). Diesel is also used during sawing, drying, and planing in lumber production. Combustion of diesel releases SO_2 from sulfur in the fuel, and NO_x forms when atmospheric nitrogen reacts with oxygen under high-temperature combustion conditions (Hebbar, 2014). Avoiding these emissions through reuse or reprocessing results in large Module D AP benefits.

The EP impact category reflects contributions of nitrogen- and phosphorus-containing compounds to eutrophication, including algal blooms and oxygen depletion in water bodies ("Eutrophication Potential," n.d.). Combustion of lignite coal in the fossil-powered grid contributes to eutrophication through phosphate leaching during waste treatment of lignite spoil. Avoiding these emissions results in the incinerate scenario having the largest Module D EP benefit.

The ODP impact category reflects depletion of the stratospheric ozone layer through emissions of halocarbons, bromides, and chlorofluorocarbons (CFCs) ("Ozone Depletion Potential," n.d.). In virgin GLT manufacturing, lumber production contributes most to ODP, largely due to coal-powered processes at sawmills (Puettmann, 2025a). Coal is used to generate electricity for sawing, drying, and planing, and to produce kraft paper used in lumber packaging. Coal combustion releases chlorine-containing compounds—particulate chloride, hydrogen chloride, and organic chlorides (especially chloromethane)—that contribute to ODP (Jin et al., 2025). Avoiding these emissions through reuse and reprocessing yields large Module D ODP benefits.

The SFP impact category reflects contributions of atmospheric compounds to photochemical ozone (smog) formation, including SO_2, volatile organic compounds (VOCs), and NO_x ("Photochemical Ozone Creation Potential," n.d.). These compounds react with ultraviolet radiation to form smog. Diesel-powered machinery used in forestry and lumber production emits SO_2 and NO_x, and avoiding these emissions results in the reuse and reprocess scenarios having the largest Module D SFP benefits.

C&D Materials

The Module D LCIA results of displacing virgin products with all recoverable C&D materials in MT1 and the Concrete Baseline are shown in

Table 37. The results are reported as emission factors per kilogram of recovered materials.

Table 37. Displaced impacts from virgin product manufacturing when construction and demolition (C&D) waste materials are reused, reprocessed, recycled, or incinerated at the end-of-life (EOL).

Material	EOL Scenario	GWP_{FOSSIL} (kg CO_2e/kg)	AP (kg SO_2e/kg)	EP (kg Ne/kg)	ODP (kg $CFC\text{-}_{11}e$/kg)	SFP (kg O_3e/kg)
MTP	Reprocess	-2.91E-01	-1.83E-03	-6.58E-04	-2.47E-08	-5.73E-02
MTP	Reuse	-2.91E-01	-1.83E-03	-6.58E-04	-2.47E-08	-5.73E-02
MTP/Wood	Recycle	-7.84E-02	-6.90E-04	-1.75E-04	-7.80E-10	-1.97E-02
MTP/Wood	Incinerate	-4.56E-01	-1.28E-03	-2.27E-03	-4.84E-09	-1.33E-02
Aluminum	Recycle	-8.27E+00	-5.43E-02	-7.69E-05	-3.73E-10	-4.21E-01
Concrete	Recycle	-2.30E-02	-1.19E-04	-4.67E-05	-2.31E-09	-1.99E-03
Gypsum	Recycle	-5.00E-02	-3.32E-04	-1.75E-04	-3.96E-10	-8.83E-03
Steel/Rebar	Recycle	-1.00E+00	-3.25E-03	-4.81E-03	-4.88E-09	-4.38E-02

The materials with the largest Module D emission factors are aluminum in the GWP_{FOSSIL}, AP, and SFP impact categories, and steel/rebar in the remaining categories. These metals have particularly large Module D impacts on a per-kilogram basis because, unlike the other materials, recycled steel and aluminum displace not only raw material extraction (A1) but also energy-intensive primary production routes. For these two materials, Module D impacts are calculated as the difference between secondary production (using scrap) and primary production (using virgin ores). Examining each material reveals the drivers behind the large Module D impacts in each category.

Secondary aluminum production consumes roughly 5% of the energy required for primary production and avoids the calcination step, which heats aluminum hydroxide slurry to 1,000–1,100°C for several hours (Walker, n.d.). This contributes to Module D GWP_{FOSSIL} impacts because secondary production avoids combustion of GHGs for electrical power. The displaced electricity from the grid is partially coal-powered, resulting in reductions in the other four impact categories. Coal combustion emits chloromethane and nitrogen oxides (NO_x), contributing to ODP, AP, and SFP. Upstream coal mining leaches phosphate ions to water, contributing to EP.

Similarly, primary production of rebar and steel involves converting iron ore into molten steel by heating and oxygenating it in a blast oxygen furnace. Secondary steel/rebar production involves melting steel scrap in an electric arc furnace, which uses electrical currents rather than coal combustion. Coal displacement contributes to the Module D GWP_{FOSSIL}, EP, and ODP impacts. Displacement of iron ore contributes to the remaining Module D categories (AP and SFP), since iron ore is transported via oceanic freighters powered by heavy fuel oil.

Unlike the metals, virgin gypsum powder and recycled gypsum powder do not use different production methods. The environmental savings come instead from displacing virgin gypsum production, which relies heavily on diesel and blasting explosives during bauxite ore mining. These fuels and materials are the largest contributors to virgin gypsum production across all impact categories, and their displacement drives the Module D impacts for gypsum.

Lastly, recycled concrete aggregate displaces virgin aggregate such as gravel and limestone. Because limestone represents the majority of the virgin aggregate mix, its displacement accounts for most of the Module D impacts in all categories. Limestone mining consumes diesel and blasting explosives, contributing to AP and SFP. After mining, limestone is milled using electricity; upstream coal use for grid electricity contributes to GWP_{FOSSIL}, AP, EP, and ODP. Milling also consumes heavy fuel oil, contributing to ODP.

Understanding how displaced processes contribute to each material's Module D impacts is important for the whole-building analysis. The major contributing materials to each building's total Module D impacts are explored in the "Whole Building Benefits" section, enabling a deeper understanding of how displaced virgin production compares between material types

Biogenic Carbon Storage

ISO Accounting Method

The biogenic carbon flows for MTP on a 100-year time horizon in each EOL scenario under the ISO 13391 Accounting Method are reported in Table 38 through Table 42. For the circular economy scenarios—reuse, reprocess, and recycle—flows are reported for the original product system (for the first 75 years) and the recovered product system (for the remaining 25 years). For the disposal scenarios—incinerate and landfill—only the first product system flows are reported, since the disposed MTP does not cross the system boundary. Biogenic carbon mass losses occur during lumber manufacturing (A1) and MTP manufacturing (A3); for simplicity, the negative flows are reported starting in A3 once the product leaves the factory.

Landfilling is the only scenario that retains biogenic carbon in permanent storage (i.e., longer than 100 years) in the first product system, resulting in a net negative $GWP_{BIOGENIC}$ value. However, the biogenic carbon stored in recovered MTP carried over into the second product system produces net negative $GWP_{BIOGENIC}$ values for the reuse, reprocess, and recycle scenarios. The landfill scenario had the lowest and largest $GWP_{BIOGENIC}$ value, followed by the reprocess scenario.

Table 38. Biogenic carbon inventory results for 1 kilogram of reused mass timber panels (MTP), by life cycle phase, across two product systems. The system boundary between the two systems is denoted with the thick black line.

$GWP_{BIOGENIC}$ – Reuse	**A3**	**C3**	**C4**	**A1**	**Total**
BCRP (kg CO_2e)	-1.83	-	-	-1.37	-3.20
BCEP (kg CO_2e)	-	1.37	0.46	-	1.83
Sum					-1.37

Table 39. Biogenic carbon inventory results for 1 kilogram of reprocessed mass timber panels (MTP), by life cycle phase, across two product systems. The system boundary between the two systems is denoted with the thick black line.

$GWP_{BIOGENIC}$ –Reprocess	**A3**	**C3**	**C4**	**A1**	**Total**
BCRP (kg CO_2e)	-1.83	-	-	-1.53	-3.36
BCEP (kg CO_2e)	-	1.53	0.30	-	1.83
Sum					-1.53

Table 40. Biogenic carbon inventory results for 1 kilogram of recycled mass timber panels (MTP), by life cycle phase, across two product systems. The system boundary between the two systems is denoted with the thick black line.

$GWP_{BIOGENIC}$ - Recycle	**A3**	**C3**	**C4**	**A1**	**Total**
BCRP (kg CO_2e)	-1.83	-	-	-0.65	-2.48
BCEP (kg CO_2e)	-	0.65	1.18	-	1.83
Sum					-0.65

Table 41. Biogenic carbon inventory results for 1 kilogram of incinerated mass timber panels (MTP), by life cycle phase, in the original product system.

$GWP_{BIOGENIC}$ - Incinerate	A3	C3	C4	Total
BCRP (kg CO_2e)	-1.83	-	-	-1.83
BCEP (kg CO_2e)	-	-	1.83	1.83
Sum				0.00

Table 42. Biogenic carbon inventory results for 1 kilogram of landfilled mass timber panels (MTP), by life cycle phase, in the original product system.

$GWP_{BIOGENIC}$ - Landfill	A3	C3	C4	Total
BCRP (kg CO_2e)	-1.83	-	-	-1.83
BCEP (kg CO_2e)	-	-	0.18	0.18
Sum				-1.65

Radiative Forcing Accounting Method

Carbon storage benefits under the Radiative Forcing method for 1 kg of MTP were estimated over a 100-year time horizon (

Table 43). The ISO method storage values are also shown for comparison. The analysis assumes that MTP remains in service for 75 years before entering one of five EOL scenarios. Storage benefits for the remaining 25 years were then calculated, accounting for mass losses during waste processing in each scenario. As with the ISO method, the resulting $GWP_{BIOGENIC}$ values from storage can be added to the GWP_{FOSSIL} emissions from the product life cycle to calculate GWP_{TOTAL}. Figure 36 illustrates the timing of biogenic carbon emissions under each scenario.

Table 43. Biogenic global warming potential ($GWP_{BIOGENIC}$) calculated with the ISO and Radiative Forcing carbon accounting methods in 1 kilogram (kg) of mass timber panel (MTP) waste over a 100-year time horizon, in different end-of-life (EOL) scenarios.

EOL Scenario	$GWP_{BIOGENIC}$ (kg CO_2e/kg MTP)- ISO	$GWP_{BIOGENIC}$ (kg CO_2e/kg MTP)- Radiative Forcing
Reuse	-1.37	-1.74
Reprocess	-1.53	-1.77
Recycle	-0.65	-1.59
Incinerate	0.00	-1.46
Landfill	-1.65	-1.79

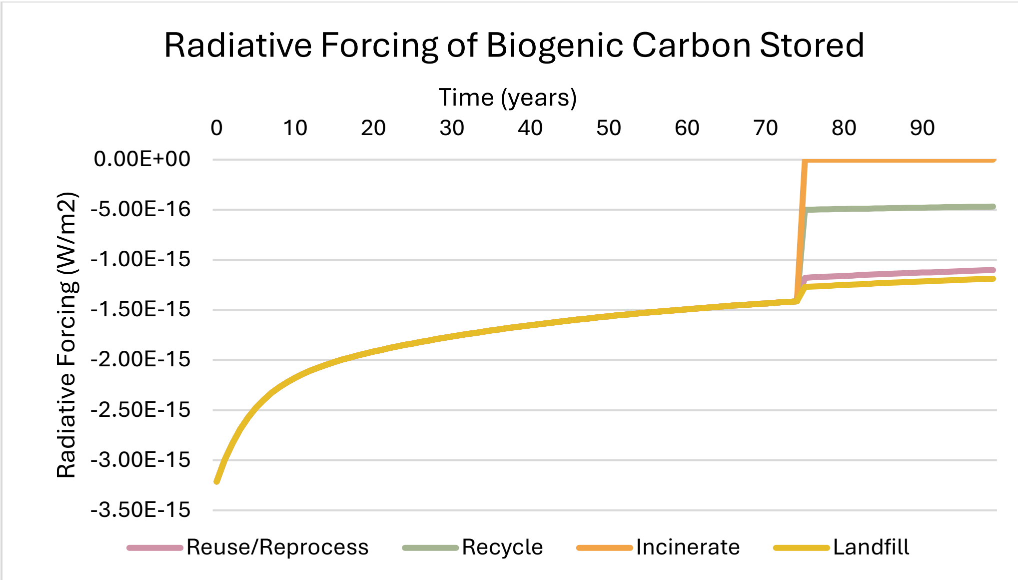

Figure 36. Radiative forcing benefits of storing biogenic carbon in mass timber panel (MTP) waste, under different end-of-life (EOL) scenarios.

Similar to the ISO accounting method, the Radiative Forcing method still shows landfilling as providing the largest carbon storage benefit. However, the relative differences between scenarios narrow considerably. Under the Radiative Forcing approach, reuse stores only 3% less carbon than landfilling, reprocess stores 1% less, and recycling stores 11% less. By comparison, the ISO method shows much larger gaps: 17% less for reuse, 7% less for reprocess, and 61% less for recycling.

Since the Radiative Forcing method applies a discounting factor over time based on the atmospheric decay of CO_2, it places greater weight on carbon stored in the early years of a product's life and less weight on carbon retained later in the time horizon, such as after EOL. This reduces the relative advantage of landfill's long-term storage and deemphasizes differences that occur late in the 100-year window. As a result, the method strengthens the case for salvaging and recovering wood products within a circular economy.

Concrete Carbonation

Summing the total cradle-to-grave CO_2 uptake for concrete yields a value of 0.035 kg CO_2 per kg concrete (83.17 kg CO_2 per m^3) in both the landfill and recycle EOL scenarios; including Module D yields an A1-C4+D value of 0.039 kg CO_2 per kg recycled concrete (93.17 kg CO_2 per m^3) (Table 35). This is comparable to a different estimate of concrete carbonation potential through next use, which reported an A1-C4+D value of 0.031 kg CO_2 per kg concrete (Van Roijen et al., 2024).

Table 44. CO_2 uptake from carbonation of CCM in concrete in each phase of the life cycle under two end-of-life scenarios: landfill and recycle.

Life Cycle Phase	Landfill Carbonation (kg CO_2/kg concrete)	Recycle Carbonation (kg CO_2/kg concrete)
A1-B5 (production to use)	3.05E-02	3.05E-02
C3 (waste processing)	4.17E-03	4.17E-03
C4 (waste disposal)	0.00E+00	0.00E+00
D (secondary use)	0.00E+00	4.17E-03
Total (A1-C4+D)	*3.47E-02*	*3.88E-02*

Whole Building Benefits

The Module D, biogenic carbon storage, and concrete carbonation emission factors were applied to the material quantities in both case study buildings under a BAU scenario. For MT1, an Optimistic scenario was also evaluated in which all structural MTP was reused. Both ISO and Radiative Forcing methods were applied to MT1's biogenic carbon.

Across all building–scenario combinations, the Concrete Baseline has the largest Module D GWP_{FOSSIL} and EP benefits and the second-largest AP, SFP, and ODP benefits (Figure 37). These results are driven primarily by displacement of virgin steel and rebar (Figure 38). Recycling steel avoids coal use in primary steelmaking, reducing GWP_{FOSSIL} and EP, and avoids upstream iron ore transport, reducing AP, SFP, and ODP.

Under the Optimistic scenario, MT1 has the largest AP, ODP, and SFP Module D benefits (Figure 37). Virgin MTP displacement dominates these categories (Figure 38). Reusing MTP avoids diesel use in forestry operations and coal-powered electricity in manufacturing, producing large AP, SFP, and ODP reductions. Steel contributes similarly to AP, explaining the comparable AP results between MT1-Optimistic and the Concrete Baseline.

Under BAU, MT1 has the lowest Module D benefits because most MTP is landfilled and does not displace virgin products. This highlights the influence of waste management: diverting MTP from the landfill to be reused shifts the whole-building comparison, with MT1 outperforming the Concrete Baseline in AP, ODP, and SFP when reuse is maximized. However, the Concrete Baseline retains larger GWP_{FOSSIL} and EP benefits due to the high coal intensity of virgin steel production.

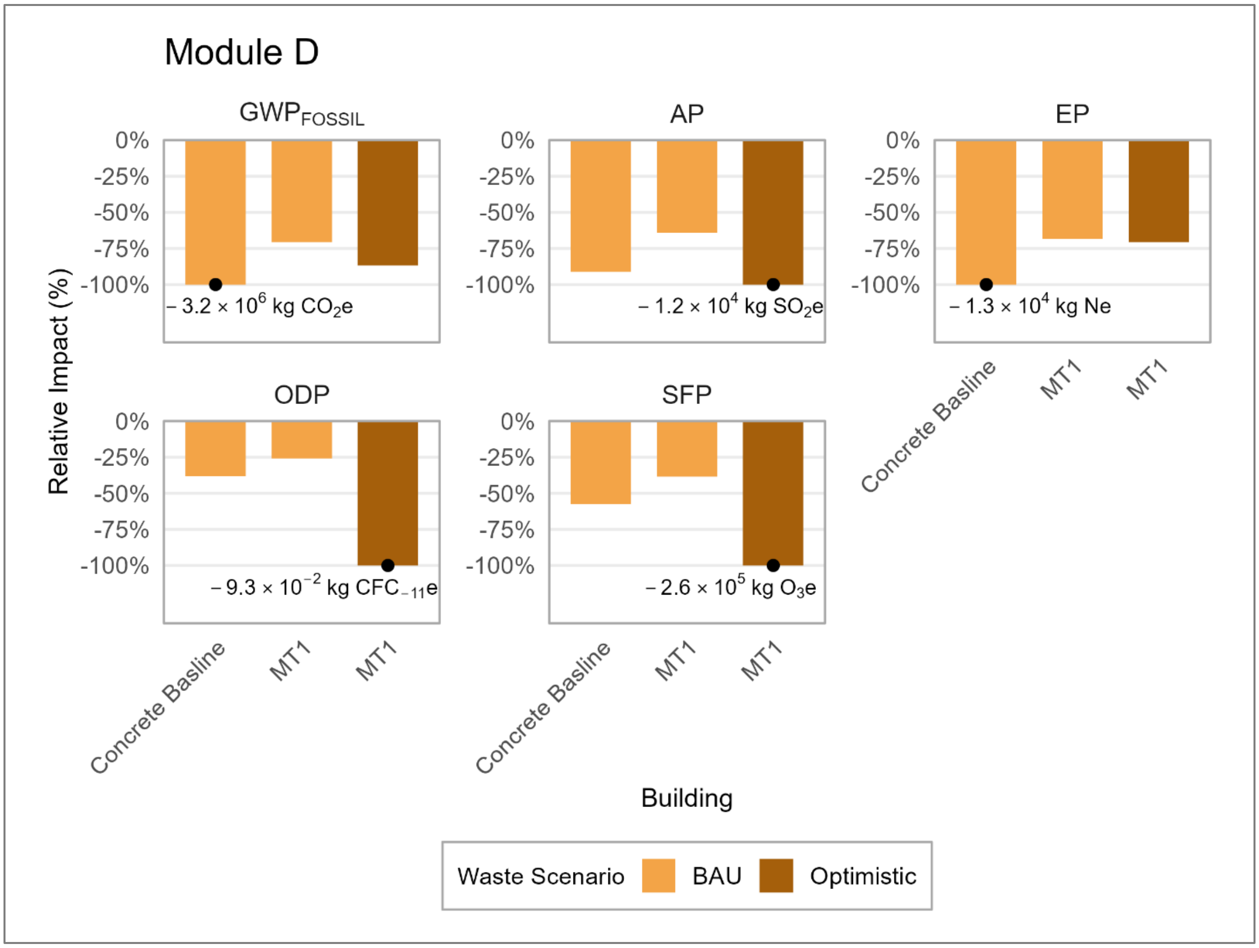

Figure 37. Relative Module D impacts of avoiding virgin material production when construction and demolition (C&D) materials are reused, recycled, or incinerated at their end-of-life in the MT1 and Concrete Baseline case study buildings. Results are normalized against the building/waste scenario combination with the highest impact in each impact category.

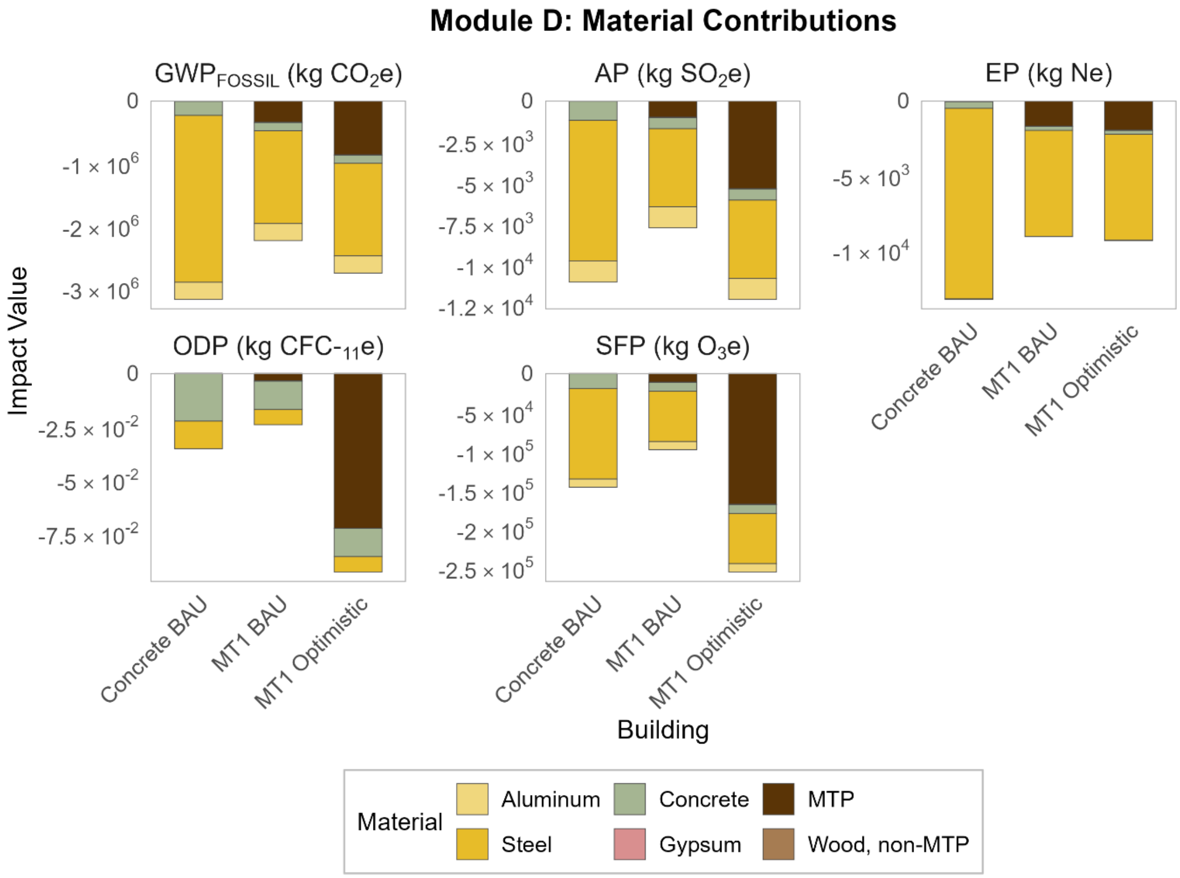

Figure 38. Module D impacts of avoiding virgin material production when construction and demolition (C&D) materials are reused, recycled, or incinerated at their end-of-life in the MT1 and Concrete Baseline case study buildings. The impacts of individual materials in each building/waste scenario combination is shown.

Under the BAU ISO scenario, MT1 stores 307% more carbon than the Concrete Baseline (Figure 39). MT1 contains roughly 26 times more wood by mass, and because the ISO method assigns the most storage to landfilling, MT1 receives large storage benefits under the BAU scenario. Although the Concrete Baseline uses nearly twice as much concrete, carbonation per kilogram of concrete is small relative to biogenic carbon stored in landfilled MTP. Thus, concrete carbonation dominates storage in the Concrete Baseline, while MTP and other wood products dominate in MT1 (Figure 40).

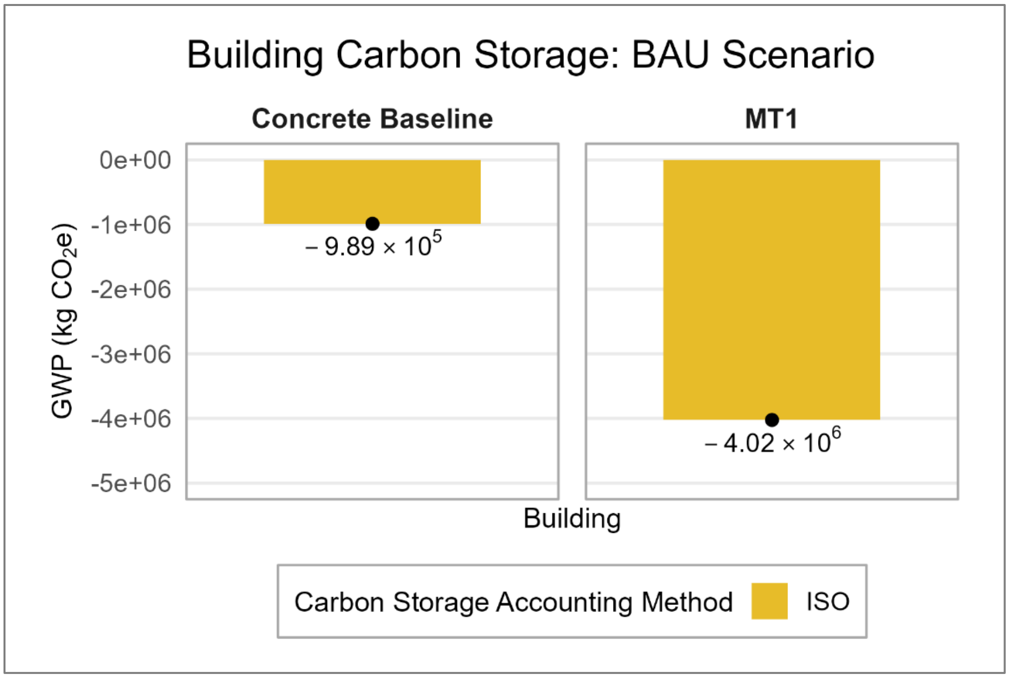

Figure 39. Total biogenic carbon storage and concrete carbonation in the MT1 and Concrete Baseline case study buildings under the Business-as-Usual (BAU) waste scenario, with the ISO carbon storage accounting method.

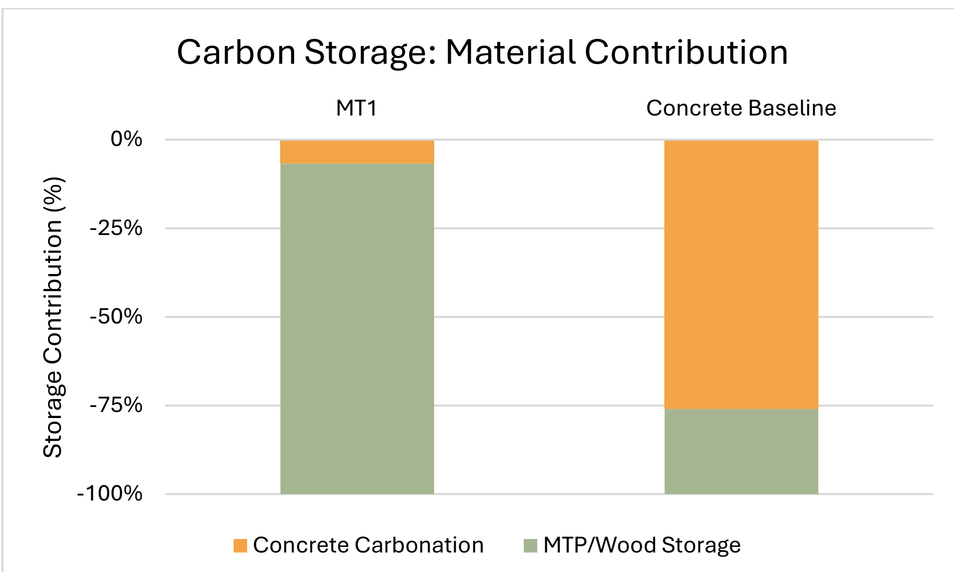

Figure 40. Contribution of individual materials (concrete, mass timber panels (MTP)/wood) to total building biogenic carbon storage and carbonation in the MT1 and Concrete Baseline case study buildings.

Differences in MT1's carbon storage across waste scenarios and accounting methods are shown in Figure 41. The Radiative Forcing method yields larger storage benefits than ISO in both scenarios: +37% in BAU and +27% in Optimistic. Despite these magnitude differences, both methods point to the same directional conclusion. The Optimistic scenario stores more carbon than BAU—11% more under ISO and 3% more under Radiative Forcing. Under BAU conditions, a portion of MTP is incinerated or recycled, both of which result in little to no carbon physically stored after EOL. Diverting this MTP to reuse in the Optimistic scenario increases the total mass of carbon retained.

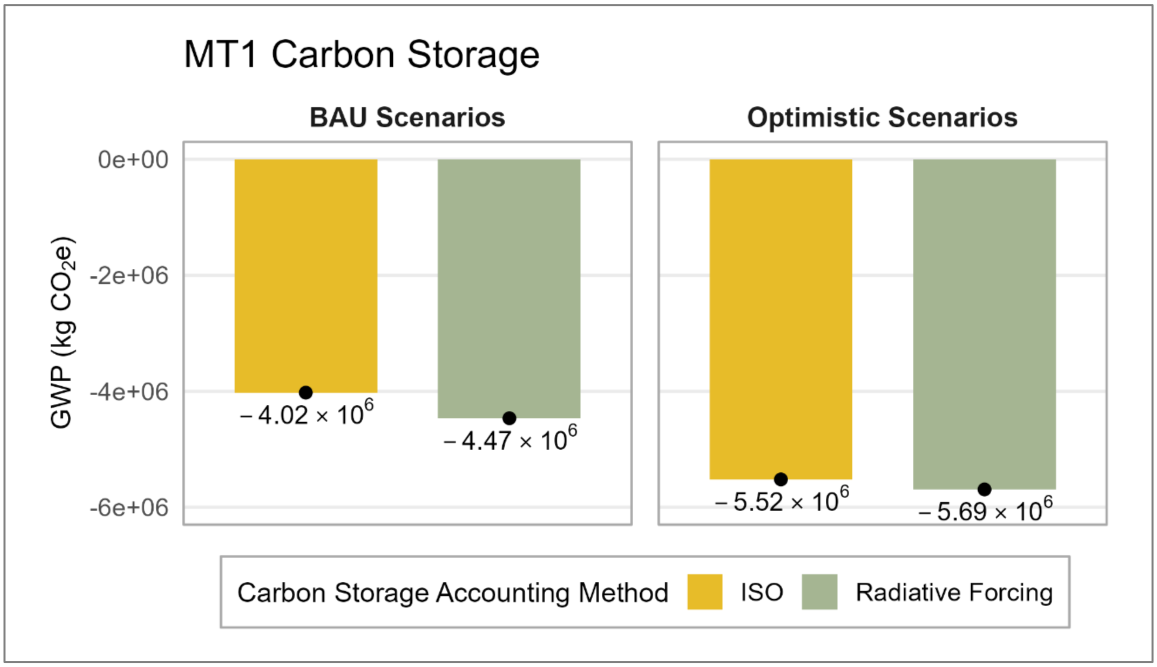

Figure 41. Total biogenic carbon storage and concrete carbonation in the MT1 case study building under the Business-as-Usual (BAU) and Optimistic waste scenarios, using difference carbon storage accounting methods.

Overall, MT1 has substantially larger combined biogenic carbon storage and carbonation benefits than the Concrete Baseline due to its higher wood content. The choice of carbon accounting method influences the total storage benefit attributed to MT1, with the Radiative Forcing method producing larger benefits because it places greater weight on early-life storage. Diverting MTP from incineration or recycling to reuse under the Optimistic scenario increases total storage under both accounting methods. The Optimistic scenario evaluated with the Radiative Forcing method yields the largest total storage benefit, reflecting the method's stronger climate valuation of the reuse pathway.

Limitations

Uncertainty in Module D results arises from assumptions about which virgin products are displaced. For MTP, the reprocess scenario assumes displacement of butcher-block countertops, but due to a lack of published data, virgin GLT production was used as a proxy; the comparability of these impacts is unknown.

In the landfill scenario, biogenic materials were assumed not to displace virgin products. Because the WARM national average mix includes no landfill gas (LFG) capture, landfill was modeled without electricity-generation benefits. As a result, Module D findings do not apply to landfills with LFG-to-energy systems, which affects conclusions about environmentally preferable EOL pathways for MTP.

For many recyclable C&D materials, Module D relied on generic LCA databases. Manufacturing impacts for displaced steel, concrete, and gypsum products were estimated using DATASMART (LTS, 2023), though U.S. industry-average EPDs would provide higher-quality data. DATASMART adapts European ecoinvent processes to U.S. conditions, but the underlying industrial inputs reflect European practices, limiting representativeness.

This limitation is evident in the steel analysis: the DATASMART dataset assumes 83% of upstream iron ore transport occurs via oceanic freighters, despite domestic iron ore production exceeding imports nearly fourfold in 2025 (USGS, 2026). As a result, Module D benefits for steel in AP and SFP may be overstated due to the assumed displacement of heavy fuel oil used in maritime transport.

Uncertainty also arises in the assumption that recycled concrete aggregate displaces virgin aggregate. Virgin aggregate was estimated as 71% limestone and 29% gravel and sand, but because the gravel–sand split was unspecified, all 29% was conservatively assigned to gravel. Future data distinguishing gravel from sand would improve accuracy.

Temporary biogenic carbon storage estimates introduce additional uncertainty. Prior studies show temporary storage can increase long-term atmospheric CO_2 once carbon is released (Levasseur et al., 2013; Korhonen et al., 2002; Kirschbaum, 2006). In this analysis, reuse, reprocess, and recycle scenarios retain up to 84% of biogenic carbon after EOL processing, with subsequent systems either emitting all carbon via incineration or storing 90% via landfilling. Whether prior findings apply to this specific sequence is unclear.

Finally, EOL carbonation of concrete depends on particle size, storage conditions, and exposure duration, all of which vary widely across waste processors. Limited empirical data exist on how these conditions influence carbonation, but the best available estimates were used (Stripple & Gustafsson, 2018).

Conclusions

This analysis quantified the Module D impacts of displacing virgin material production for multiple construction and demolition (C&D) materials across reuse, reprocess, recycle, and incinerate end-of-life (EOL) scenarios. Emission factors were developed for each material and applied to a mass timber panel (MTP) case study building (MT1) and an alternate reinforced concrete design (Concrete Baseline). A business-as-usual (BAU) waste scenario was modeled for both buildings, and an Optimistic scenario—diverting all landfilled, recycled, or incinerated structural MTP to reuse—was applied to MT1.

At the material level, Module D emission factors for MTP varied substantially across EOL scenarios. Incineration produced the largest benefits in the global warming (GWP_{FOSSIL}) and eutrophication (EP) categories due to avoided fossil-fuel electricity. Reuse produced the largest acidification (AP), ozone depletion (ODP), and smog formation (SFP) benefits because it displaces diesel use in forestry operations. Comparing all C&D materials showed that aluminum, rebar, and steel had the largest per-kilogram Module D benefits, as recycled metals avoid both virgin material extraction and energy-intensive primary production. Coal displacement drives most of steel's Module D benefits, while electricity displacement drives aluminum and concrete benefits, and diesel displacement drives gypsum benefits.

At the whole-building scale, the Concrete Baseline had the largest Module D benefits in the GWP_{FOSSIL} and EP categories, driven primarily by steel recycling. Under the Optimistic scenario, MT1 had larger Module D benefits in AP, ODP, and SFP due to MTP reuse. Diverting MTP from landfill improved MT1's overall performance relative to the Concrete Baseline, as diesel displacement during virgin MTP production yields larger AP, SFP, and ODP reductions than steel recycling on a whole-building basis.

Biogenic carbon storage and concrete carbonation assessments showed that MT1 stored more carbon than the Concrete Baseline under the BAU ISO scenario, reflecting MT1's substantially higher wood content. Comparing MT1's waste scenarios also showed that both accounting methods assign greater storage benefits to the Optimistic scenario. The Radiative Forcing method paired with the Optimistic scenario produces the largest total storage benefit, because this method places less weight on late-stage storage and therefore narrows the gap between reuse storage and the large storage attributed to landfill.

Key Findings

This chapter evaluated the Module D environmental benefits and burdens associated with substituting virgin materials when construction and demolition (C&D) waste is recovered at end-of-life (EOL). The analysis was conducted for a mass timber panel (MTP) case study building (MT1) and its concrete alternative, and carbon storage benefits were assessed using both ISO guidelines and a Radiative Forcing accounting method. Several overarching findings emerged:

1. **MTP Module D benefits were largest in the reuse and incinerate scenarios.**
   Among the reuse, reprocess, recycle, and incinerate scenarios, reuse and incineration produced the greatest impact reductions. Reuse avoids diesel consumption in virgin forestry operations, while incineration avoids fossil-fuel electricity generation in the U.S. grid.
2. **Fossil energy displacement contributes to Module D impacts for other C&D waste.**
   Recycled metals—aluminum, steel, and rebar—showed the largest per-kilogram Module D benefits because they displace both virgin material extraction and energy-intensive primary production. Concrete recycling avoids diesel and explosives used in limestone mining, while gypsum recycling avoids diesel used in virgin gypsum production.
3. **Carbon-storage accounting methods determine which EOL scenario appears most climate-beneficial.**
   Under ISO 21930 and 13391, the landfill scenario receives the largest carbon storage credit. Under the Radiative Forcing method, however, the gap between reuse/reprocess storage and landfill storage narrows because this method places greater weight on early-stage storage. At the whole-building level, MT1 stores more carbon in the Optimistic scenario under both accounting methods, with the Radiative Forcing method yielding the largest total storage benefit due to the diversion of MTP from incineration or recycling into reuse.

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Chapter 4: Environmental Impacts and Benefits across a Cradle-to-Grave Scope

Introduction

To date, only two life cycle assessment (LCA) studies have assessed the cradle-to-grave impacts of mass timber panel (MTP) buildings in the United States (U.S.) (Chen et al., 2020; Liang et al., 2021). Limitations in these studies highlight persistent gaps in understanding the comparative whole-life performance of U.S.-produced mass timber and concrete buildings, particularly in the use and end-of-life (EOL) phases. These gaps were addressed in the previous three chapters of this dissertation to provide comprehensive LCA data on each life cycle phase for MTP and other C&D materials.

However, there are no U.S.-specific analyses that compare different EOL pathways for MTP on a whole-building scale. Beyond substitution benefits, retaining MTP in the circular economy through reuse or recycling can provide additional environmental advantages. These include improved resource efficiency, reduced burdens from raw-material extraction, increased wood use, longer carbon storage in harvested wood products, reduced waste-disposal costs, and local job creation (Lazaridou et al., 2021). Diverting MTP from the landfill to be reused can improve its climate performance in the EOL phase when temporary carbon storage benefits are included (Bjarvin et al., 2025), though its influence on the full life cycle for a building is unknown. This underscores the need for a comparison of different waste scenarios and carbon storage accounting methods for MTP and reinforced concrete construction in a cradle-to-grave scope.

Research Objectives

To address the gaps in U.S. mass timber LCA research, this analysis conducts a comprehensive cradle-to-grave whole building LCA (WBLCA) of two case study buildings: Google's MT1 mass timber building and its alternative concrete design ("Concrete Baseline"). The environmental impacts of production/construction (A1–A5), use (B1–B4), EOL treatment (C1–C4), virgin product substitution (Module D), and carbon storage benefits were previously assessed in Chapters 1-3. The whole-building results of each module are synthesized to estimate cradle-to-grave impacts of each building.

To evaluate how increased MTP reuse influences cradle-to-grave impacts, two waste scenarios are modeled: (1) demolition of both buildings using standard waste distribution rates, and (2) selective deconstruction of MT1 with increased reuse of structural MTP. As this chapter synthesizes the findings of the previous three chapters in this dissertation, the "Conclusions" and "Key Findings" sections are presented for the entire dissertation at the end of the chapter.

Methods

Goal & Scope

This LCA was conducted in accordance with ISO 14040, ISO 14044, and ISO 21930 (ISO, 2006a-b; ISO, 2017). All impacts were modeled in SimaPro v9.5 using the TRACI characterization method (Bare, 2012). The functional unit for this analysis is one building.

The environmental burdens and benefits associated with displacing virgin products using reclaimed construction and demolition (C&D) waste are quantified in Module D. Figure 42 illustrates the system boundary and shows how processes and materials are included in each life cycle phase. While the virgin product substitution impacts are only evaluated in Module D, carbon storage benefits are evaluated starting in the production phase (A1) and ending after EOL (C3-C4).

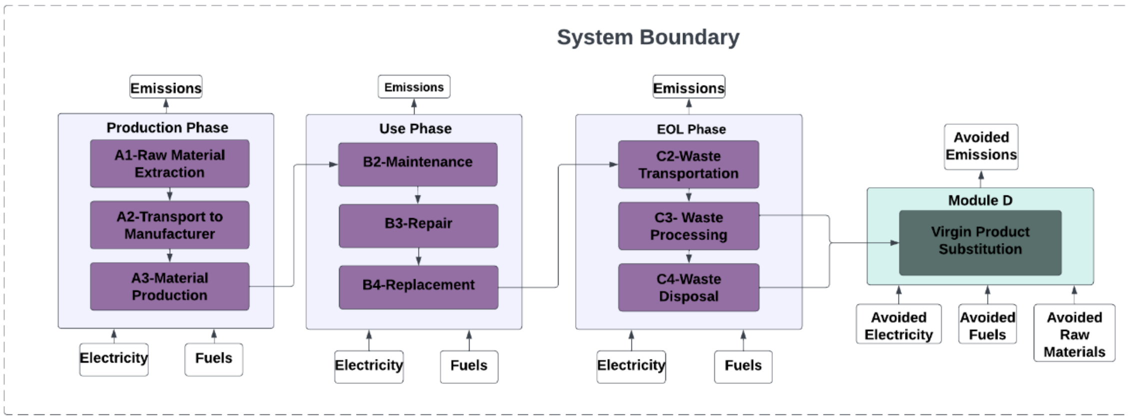

Figure 42. System boundary diagram for carbon storage and virgin product substitution by recovered construction and demolition waste.

This study evaluates the five mandatory impact categories specified in ISO 21930:2017 (Table 45). Two additional inventory parameters (biogenic carbon removals and emissions from products) were included to track biogenic carbon flows across the system boundary.

Table 45. Impact category indicators per ISO 21930 (ISO, 2017).

Impact Categories	Abbreviation	Units	Method
Global warming potential, fossil	GWP_{FOSSIL}	kg CO_2e	TRACI 2.1 V1.08
Acidification potential	AP	kg SO_2e	TRACI 2.1 V1.08
Eutrophication potential	EP	kg Ne	TRACI 2.1 V1.08
Ozone depletion potential	ODP	kg $CFC\text{-}_{11}e$	TRACI 2.1 V1.08
Smog formation potential	SFP	kg O_3e	TRACI 2.1 V1.08
Global warming potential, biogenic	$GWP_{BIOGENIC}$	kg CO_2e	TRACI 2.1 V1.08 + LCI Indicatory
Biogenic carbon removal from product	BCRP	kg CO_2e	LCI Indicatory
Biogenic carbon emission from product	BCEP	kg CO_2e	LCI Indicatory

Because recycled materials are included in the analysis, the avoided burden (0:100) allocation approach was adopted (De Wolf et al., 2020). Under this method, all burdens and benefits associated with recycling are assigned to the first life cycle of the material, rather than the secondary product. This enables explicit calculation of Module D impacts and aligns with common practice for closed-loop recycling systems such as steel and aluminum, where scrap is returned to primary production to create the same product type (World Steel Association, 2017).

Scenario Definition

Two waste scenarios were modeled for each building, with additional sub-scenarios for MT1 to evaluate biogenic carbon storage under two accounting methods:

- **Business-as-Usual (BAU):** Demolition + BAU disposal
  - BAU ISO
  - BAU Radiative Forcing
- **Optimistic:** Selective deconstruction + reuse of MTP
  - Optimistic ISO
  - Optimistic Radiative Forcing

Under the BAU scenario, both buildings are demolished and material flows follow U.S. national average C&D waste distribution rates (EPA, 2020). Under the Optimistic scenario, MT1 is deconstructed rather than demolished, enabling recovery and reuse of 100% of the structural MTP.

Two carbon accounting methods (detailed in Chapter 3) were applied to both waste scenarios to evaluate how carbon storage assumptions influence the overall GWP_{FOSSIL} results. The Concrete Baseline contains only small quantities of lumber and plywood in the enclosure and interior. For this building, only the ISO method is applied. The Radiative Forcing method is evaluated solely for MT1 to illustrate how increased MTP reuse affects carbon storage outcomes.

This structure enables two levels of comparison:

1. **MTP vs. Reinforced Concrete Construction** under the BAU ISO scenario, using consistent waste distributions and ISO carbon accounting.
2. **Deconstruction and Reuse Benefits** under BAU and Optimistic scenarios for MT1 only, comparing both carbon accounting methods.

The inclusion of both ISO and Radiative Forcing sub-scenarios further illustrates how carbon accounting choices influence the perceived climate benefits of different EOL pathways for MTP and wood products.

Synthesis of Results

This cradle-to-grave analysis integrates all life cycle modules to evaluate the cumulative environmental performance of each building. This synthesis highlights how impacts of production (A1-A5), use phase (B1-B4), and end-of-life outcomes (C1-C4), together with Module D and carbon storage, shape the overall performance of mass timber versus concrete.

Cradle-to-grave impacts for both buildings are calculated by summing impacts from all life cycle modules, reported on a whole-building level in Chapters 1-3. Information on the source of each module's results are provided in

Table 46.

Table 46. Sources of environmental impacts of different life cycle modules for two case study buildings in this dissertation.

Module	Chapter #	Table #
Production (A1-A5)	1	Figure 7
Use (B1-B4)	1	Figure 8
End-of-Life (C1-C4)	2	Figure 32
Module D	3	Figure 37
Carbon Storage[4]	3	Figure 39 Figure 41

The full life cycle impact analysis (LCIA) results for each module are reported by building and waste scenario in Appendix E.

Results & Discussion

In the previous three chapters of this dissertation, five impact categories were evaluated using the TRACI characterization method (Bare, 2012): fossil global warming potential (GWP_{FOSSIL}), ozone depletion potential (ODP), acidification potential (AP), eutrophication potential (EP), and smog formation potential (SFP). An additional impact category (biogenic global warming potential, $GWP_{BIOGENIC}$) was evaluated manually.

MT1 was evaluated under four combinations of waste scenario and carbon accounting method (BAU ISO, BAU Radiative Forcing, Optimistic ISO, Optimistic Radiative Forcing), while the Concrete Baseline was evaluated under the BAU ISO scenario. The results are presented in two separate sections:

1. **MTP vs. Reinforced Concrete Construction** under the BAU ISO scenario
2. **Reuse Benefits** under BAU and Optimistic scenarios for MT1 only

MTP vs. Reinforced Concrete Construction

The BAU ISO results for MT1 and the Concrete Baseline provide a direct comparison of mass timber and reinforced concrete construction under typical U.S. demolition and waste distribution practices (Figure 43). Under this scenario, MT1 has lower cumulative A1-C4+D impacts in all categories except eutrophication (EP). The higher EP impact for MT1 results from leaching nitrogen-containing compounds during anaerobic decomposition of landfilled MTP. In all other categories, MT1's lower impacts reflect reductions across most life cycle phases relative to the Concrete Baseline. This is illustrated by comparing GWP_{FOSSIL} impacts between each building.

In the production/construction phase (A1-A5) and the use phase (B1-B4), MT1 has a 31% and 20% lower GWP_{TOTAL}, respectively, compared to the Concrete Baseline. Combined, these phases make up 94%-97% of the cradle-to-grave (A1-C4) GWP_{TOTAL} in each building. Conversely, MT1 has a 70% higher EOL (C1-C4) impact than the Concrete Baseline, since landfilling MTP in the BAU scenario generates methane emissions. However, since the EOL phase has a minimal contribution (3%-6%)

[4] This includes both biogenic carbon storage and concrete carbonation.

to each building's cradle-to-grave GWP_{TOTAL}, the overall effect is a net reduction in cradle-to-grave GWP_{TOTAL} for MT1.

A comparison of the climate-mitigating effects of Module D product substitution and carbon storage reveals further evidence of improved environmental performance for MTP construction. MT1 has a 30% smaller Module D GWP_{TOTAL}, which is worse for climate performance since Module D impacts are reported as negative. This is because the majority of its primary structural component (MTP) is landfilled, which generates no virgin product substitution benefits. However, MT1 has a 307% larger $GWP_{BIOGENIC}$ from carbon storage, since it uses more wood products than the Concrete Baseline. Together, the climate benefits of Module D and carbon storage offset 43% of MT1's GWP_{FOSSIL}, whereas only 22% of the Concrete Baseline's GWP_{FOSSIL} is offset.

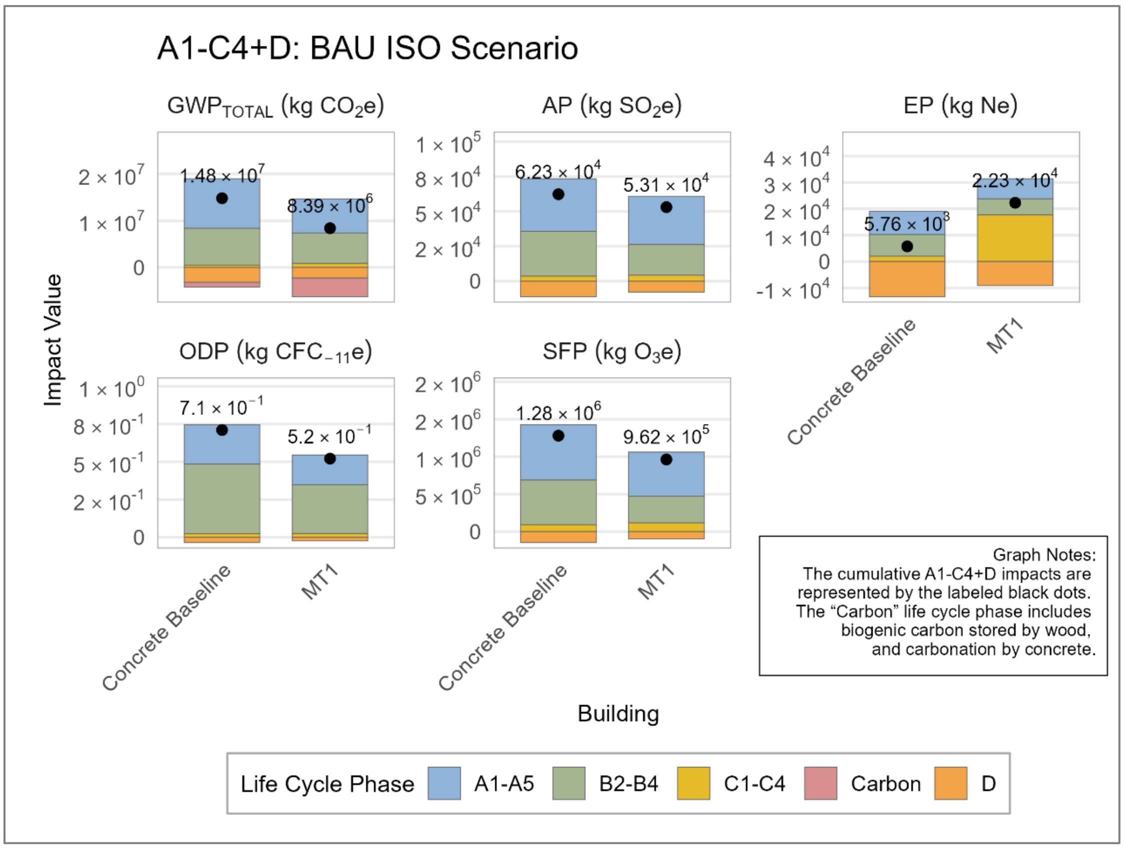

Figure 43. Cradle-to-grave (A1-C4+D) environmental impacts for the MT1 and Concrete Baseline case study buildings under the Business-as-Usual (BAU) waste scenario, using the ISO carbon storage accounting method.

Reuse Benefits

Comparing the results of the four waste scenario combinations for MT1 enables a deeper understanding of potential reuse benefits for MTP. This is achieved by comparing the BAU and Optimistic results under each carbon accounting method.

Using the ISO carbon storage method, the total GWP_{TOTAL} impact for MT1 decreases when shifting from the BAU to the Optimistic scenario (Figure 44). This occurs because the ISO method assigns the most carbon storage benefits to landfilling, and substantially less to other EOL scenarios. In the

Optimistic scenario, MTP is primarily diverted from the landfill to reuse, though a portion of MTP is also diverted from incineration and recycling. When structural MTP is reused instead of incinerated or landfilled, the carbon storage benefit attributed to the MTP increases, since the incinerate and recycle scenarios have the lowest amount of storage after EOL, whereas the reuse scenario has the second-highest total storage.

All other impact categories also decrease under the Optimistic scenario, with the largest reduction in EP due to avoiding landfill-related nitrogen leaching. Acidification (AP) and smog formation (SFP) decrease to a lesser extent because MTP reuse requires more diesel during waste processing than landfilling. These results suggest that deconstruction and reuse can decrease a building's impacts across the entire life cycle in all five impact categories, though more efficient deconstruction methods and cleaner energy sources during waste processing would expand these reductions.

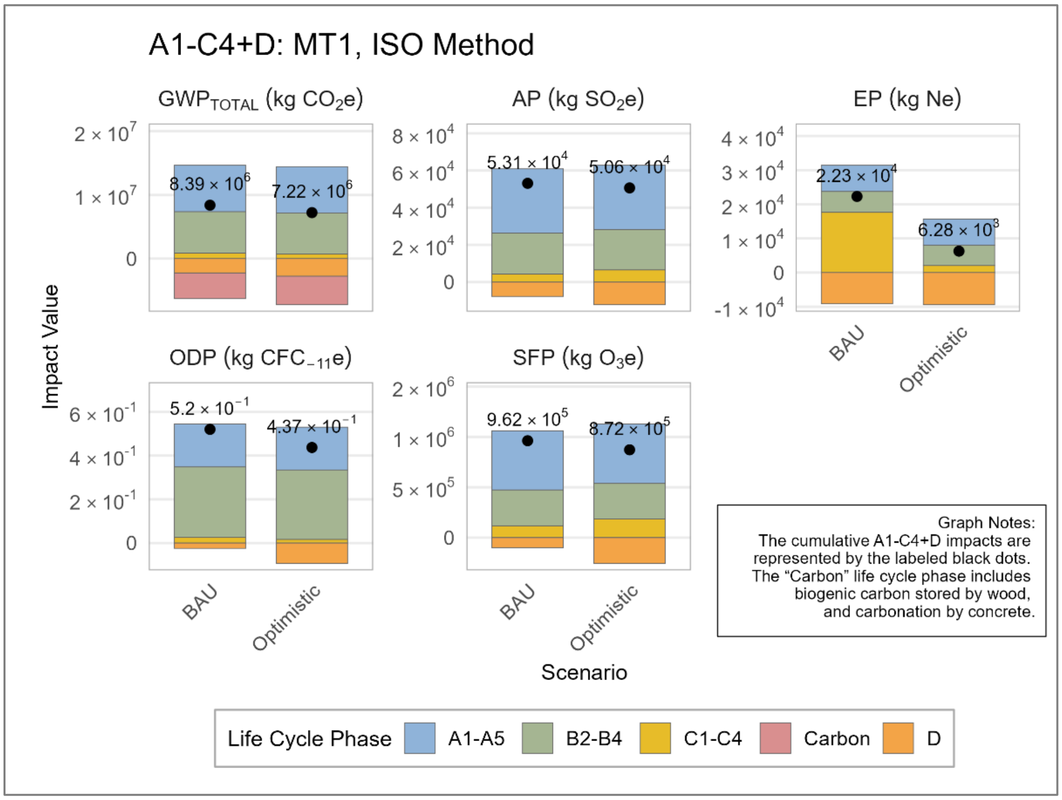

Figure 44. Cradle-to-grave (A1-C4+D) environmental impacts for the MT1 case study building under different waste scenarios, using the ISO carbon storage accounting method.

Using the Radiative Forcing method instead of the ISO method yields the same the GWP$_{TOTAL}$ conclusions, with the Optimistic scenario demonstrating lower impacts (Figure 45). Diverting MTP waste from incineration or recycling to reuse in the Optimistic scenario yields larger storage benefits under both accounting methods, though the Radiative Forcing method yields the largest total storage because it places more weight on early-life storage than EOL storage. As such, the storage differences between each EOL scenario are smaller compared to the ISO method. Reuse is credited with 17% less storage than landfill under ISO, compared to 3% less storage under Radiative Forcing, demonstrating the climate valuation of the reuse scenario under the Radiative Forcing method. This discrepancy highlights the need for further consideration of dynamic radiative forcing methods in ISO guidelines, which are currently under development ("ISO/AWI TR 14082," n.d.). These results suggest that MTP reuse can further decrease building embodied carbon across the entire life cycle if alternative carbon storage accounting methods to ISO are used.

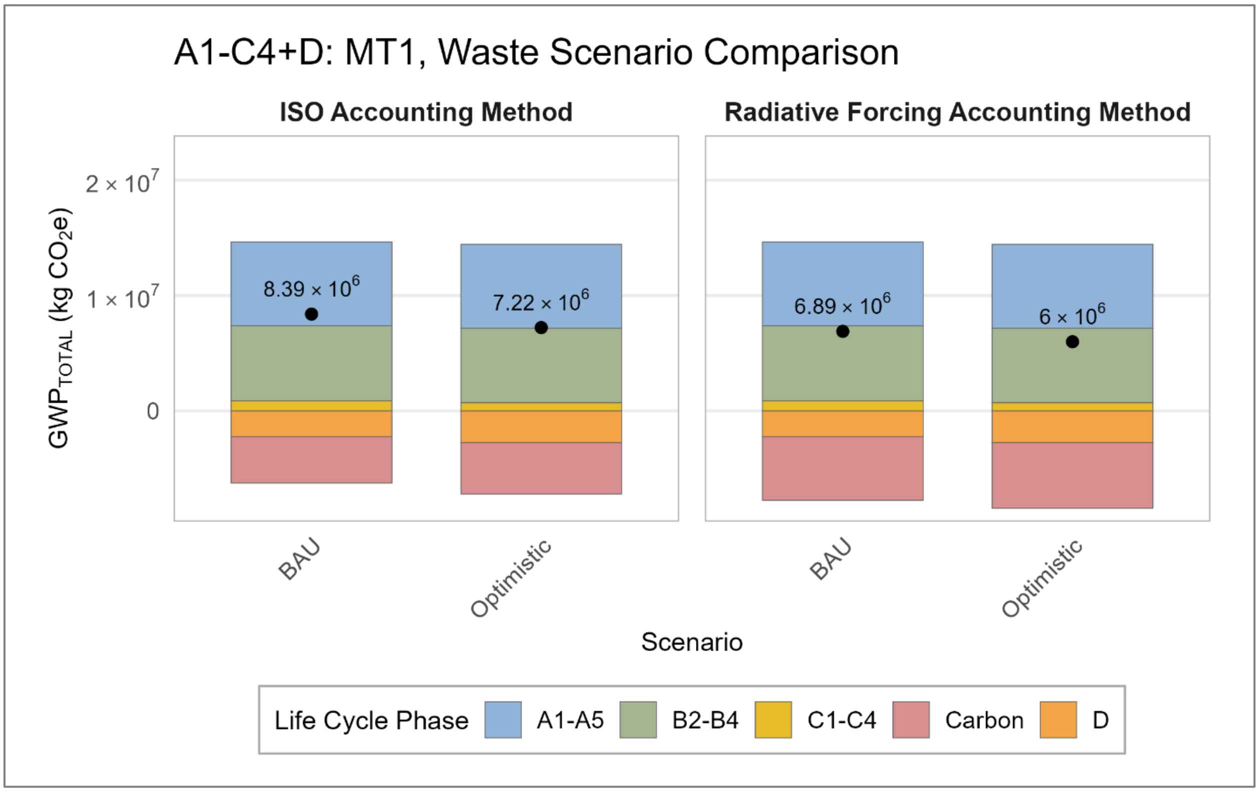

Figure 45. Cradle-to-grave (A1-C4+D) environmental impacts for the MT1 case study building under different waste scenarios and carbon storage accounting methods.

A final comparison of the cradle-to-grave results is done to compare the Concrete Baseline results to MT1 under all four possible waste scenario and carbon accounting combinations. MT1 has lower cradle-to-grave impacts than the Concrete Baseline in the GWP_{FOSSIL}, AP, ODP, and SFP categories, regardless of waste scenario or carbon accounting method (Figure 46). Reusing MTP in the Optimistic scenario also reduces MT1's EP impact to levels comparable to the Concrete Baseline, whereas the MT1 BAU scenario has much higher EP impacts, due to the landfilling of MTP under BAU practices. In all impact categories, the Concrete Baseline has the highest impacts, followed by MT1 BAU, and then MT1 Optimistic, suggesting that reusing MTP yields the lowest cradle-to-grave impacts on a whole-building level.

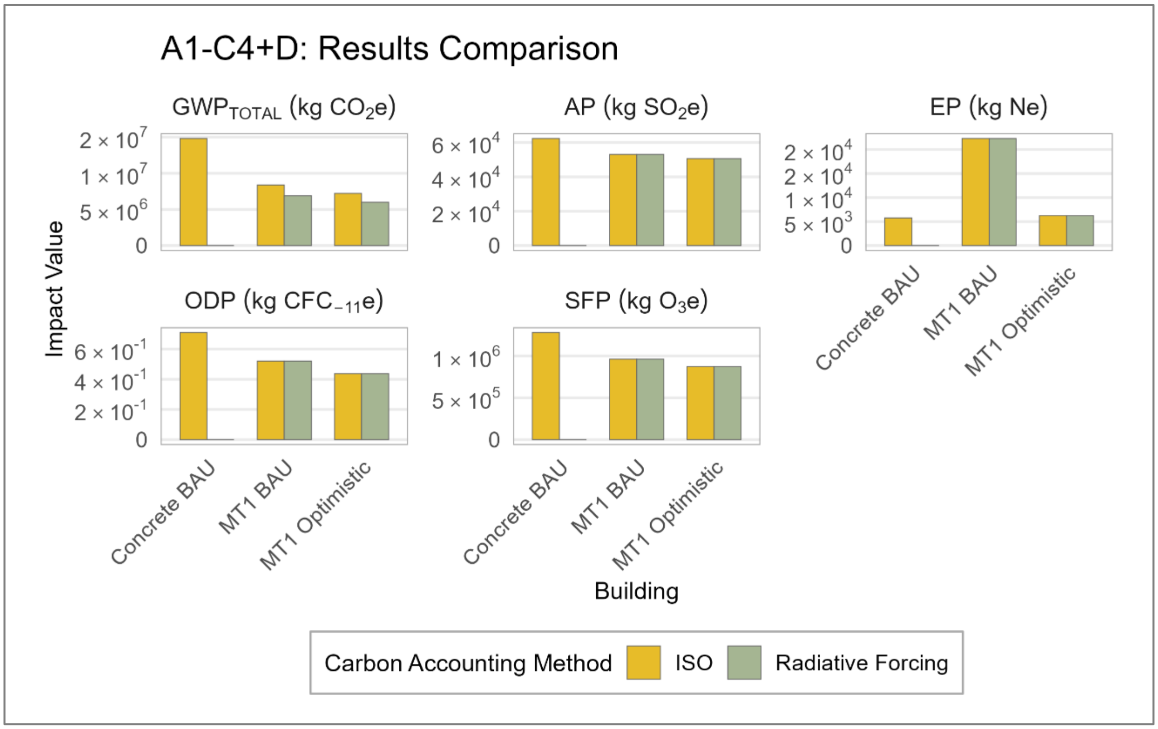

Figure 46. Cumulative cradle-to-grave (A1-C4+D) environmental impacts for the MT1 and Concrete Baseline case study buildings under different waste scenarios and different carbon storage accounting methods.

Under the Optimistic scenario, the MT1 building achieved a 52–61% reduction in cradle-to-grave GWP_{TOTAL} (depending on the carbon storage accounting method), along with reductions of 22% in AP, 8% in EP, 41% in ODP, and 35% in SFP relative to the Concrete Baseline. Under the BAU scenario, MT1 still showed substantial improvements in most categories—45–55% lower GWP_{TOTAL}, 18% lower AP, 29% lower ODP, and 28% lower SFP—but exhibited a 263% increase in EP due to nitrogen leaching from landfilled MTP. Overall, diverting MTP from landfill to reuse at end-of-life further enhances the cradle-to-grave environmental performance of mass timber construction across all impact categories when compared to reinforced concrete.

These results align with the aforementioned existing North American WBLCA studies on MTP (Liang et al., 2021; Chen et al., 2020; Kumar et al., 2024). In a 12-story comparison of mass timber and concrete, Liang et al. (2021) reported reductions for mass timber across most impact categories, ranging from -0.4% (ODP) to -11.6% (GWP_{TOTAL}). Their cradle-to-grave system boundary included operational energy and water use (B6–B7) but excluded waste transport, processing, and disposal (C4), which likely contributes to the smaller magnitude of reductions relative to those found in this dissertation. Their base case assumed 30% of MTP was landfilled, with the remainder reused or recycled. In a sensitivity scenario with 100% landfill, the GWP_{TOTAL} benefit of mass timber dropped to 8%. This pattern reinforces the conclusion of this research: reducing landfill disposal of MTP and increasing reuse substantially improves the climate performance of mass timber buildings, even when operational energy is included.

Chen et al. (2020), analyzing the same building, found similarly modest reductions—5.9% lower A–D GWP_{TOTAL} for mass timber relative to concrete when operational energy was included. When operational energy was excluded, however, the reductions increased to -69.5%, closely matching the magnitude of GWP_{TOTAL} reductions reported in this dissertation. Similarly, Kumar et al. reported reductions for MTP between -19% (AP) and -62% (EP) for all five impact categories when operational energy was not considered. Liang et al. (2021) emphasize that operational energy efficiency is critical for maximizing the environmental benefits of mass timber construction.

Conclusions

This dissertation evaluated the cradle-to-grave performance of a mass timber building (MT1) and an alternative reinforced concrete design (Concrete Baseline) under different waste scenarios. Environmental impacts were assessed across all life cycle phases—production and construction (A1-A5), use (B1-B4), and end-of-life (C1-C4)—along with the benefits of displacing virgin material production in Module D, biogenic carbon storage, and concrete carbonation for each building.

The impacts of each life cycle phase were summed to calculate the cumulative cradle-to-grave (A1-C4+D) impacts for each building under different assumptions about waste distribution and carbon storage accounting. In the BAU scenario, both buildings were assumed to be demolished, and C&D waste was distributed to end-of-life pathways based on U.S. national average waste characterization data. In the Optimistic scenario, MT1 was deconstructed to enable recovery and reuse of structural MTP. For MT1, two carbon storage accounting methods were applied to both waste scenarios: the ISO method (ISO 21930) and the Radiative Forcing method, which discounts the climate benefit of stored biogenic carbon over time. This resulted in four scenario combinations for MT1: BAU ISO, BAU Radiative Forcing, Optimistic ISO, and Optimistic Radiative Forcing.

Combining all life cycle phases to calculate A1-C4+D impacts showed that MT1 had lower cradle-to-grave impacts than the Concrete Baseline in most categories under the BAU ISO scenario, driven by lower production, construction, and use phase impacts. The Concrete Baseline had lower EP impacts because landfilled MTP leaches nitrogen during biodegradation. Comparing MT1's BAU and Optimistic scenarios under both carbon accounting methods showed that reusing MTP resulted in lower cradle-to-grave impacts in all categories; the Radiative Forcing method combined with the Optimistic scenario yielded the lowest GWP_{TOTAL} because this method places more weight on early-life storage, resulting in smaller differences in storage amongst EOL scenarios compared to the ISO method.

Overall, this analysis demonstrates that MTP construction offers environmental advantages over reinforced concrete construction across most life cycle phases and impact categories, particularly when structural MTP is recovered for reuse. While landfilling MTP under BAU conditions increases eutrophication impacts, diverting MTP to reuse substantially reduces these impacts and improves overall performance.

The choice of carbon accounting method plays a critical role in interpreting the climate benefits of MTP, particularly when comparing landfill and reuse pathways. Under accounting methods that prioritize early-life storage over EOL storage, reuse of MTP yields the lowest cradle-to-grave impacts and the greatest climate benefits. The findings also highlight the need for carbon accounting frameworks that recognize dynamic radiative forcing methods, especially as mass timber adoption accelerates in North America.

Given the substantial variation in GWP_{FOSSIL} outcomes produced by different carbon-accounting frameworks for MTP, policymakers should recognize that conclusions about the climate benefits of specific EOL pathways are highly sensitive to the chosen accounting method. As states expand Buy Clean and embodied-carbon procurement requirements, incorporating accurate and transparent EOL carbon-accounting methods will be critical to ensuring that policy decisions reflect the true climate performance of mass timber products.

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Overarching Key Findings

This dissertation evaluated the cradle-to-grave (A1–C4+D) environmental performance of a mass timber building (MT1) and an alternative reinforced concrete design (Concrete Baseline) under multiple waste and carbon accounting scenarios. Several overarching findings emerged:

1. Mass timber construction substantially reduces impacts in early life cycle phases.

Across production/construction (A1–A5) and use (B1–B4), MT1 consistently outperformed the Concrete Baseline in all impact categories. The largest reductions occurred in the production phase, where mass timber panels (MTP) displaced large quantities of reinforced concrete and steel—materials with high embodied impacts. Use-phase differences were driven entirely by enclosure materials, with MT1 benefiting from lower-impact cladding and reduced use of glass and vapor barriers.

2. During demolition, wood-framed structures require more energy than concrete structures, though deconstruction requires less energy than demolition.

Demolishing MT1 produced the highest demolition/deconstruction (C1) impacts of all scenarios due to higher diesel consumption in wood-framed demolition. Deconstruction reduced MT1's C1 impacts but remained higher than demolition of the Concrete Baseline. These results highlight the sensitivity of C1 impacts to building typology and equipment energy modeling, and demonstrate the need to develop more streamlined deconstruction processes.

3. Waste processing and disposal dominate EOL impacts.

Across both buildings, waste processing/disposal (C3–C4) contributed the majority of impacts in the EOL phase. Landfilled MTP generated high eutrophication impacts due to nitrogen leaching from degraded wood, while diesel-intensive recycling processes contributed to acidification (AP) and smog formation (SFP). The BAU scenario for MT1 had the highest overall EOL impacts; selective deconstruction and reuse reduced impacts in most categories.

4. Module D benefits vary by material and scenario.

Recycling steel in the Concrete Baseline produced larger Module D benefits in GWP_{FOSSIL} and EP than reusing MTP, reflecting steel's high substitution value. Conversely, reusing MTP generated larger Module D benefits in AP, SFP, and ozone depletion (ODP) under the Optimistic scenario. These differences underscore the importance of material-specific substitution impacts.

5. Carbon storage outcomes are influenced by the accounting method.

Under ISO 13391, landfilled wood receives the most storage benefits, with reused wood receiving 17% less storage. Under a dynamic Radiative Forcing method, reuse receives 3% less storage than the landfill, due to stronger valuations of early-life storage than EOL storage. As a result, the Optimistic scenario for MT1 has lower GWP_{TOTAL} impacts compared to the BAU scenario. This divergence demonstrates that carbon accounting choices can further demonstrate the climatic benefits of reuse.

6. Reuse of MTP reduces impacts in most categories and mitigates landfill-related burdens.

Selective deconstruction and reuse of structural MTP reduced MT1's impacts in all categories except AP and SFP, which increased due to diesel use during reuse processing. Reuse substantially reduced eutrophication impacts by avoiding nitrogen leaching from landfilled MTP.

7. Mass timber outperforms concrete in cradle-to-grave performance under most conditions.

Across all scenarios and accounting methods, MT1 had lower A1–C4+D impacts than the Concrete Baseline in GWP_{FOSSIL}, AP, ODP, and SFP. The Concrete Baseline only outperformed MT1 in EP under BAU conditions due to landfill emissions from MTP. When MTP was reused, MT1's EP impacts became comparable to the Concrete Baseline.

Taken together, these findings demonstrate that mass timber construction offers clear cradle-to-grave environmental advantages over reinforced concrete, particularly when structural MTP is recovered for reuse. Circular economy pathways—reuse, reprocessing, recycling, and energy recovery—consistently reduce environmental burdens and enhance climate performance compared to landfilling. Landfilling yields the largest carbon storage benefits of the assessed end-of-life scenarios, but the choice of carbon accounting method determines the magnitude of differences between carbon storage in each scenario, underscoring the need for frameworks that recognize the dynamic nature of biogenic carbon storage and emissions.

Appendix A

This appendix contains the life cycle inventory (LCI) data for the use phase (B1-B4) of the building life cycle, assessed in Chapter 1 of this dissertation. In this phase, materials used in the enclosure and interior components of two case study buildings are maintained (B2), repaired (B3), or replaced (B4). Material use (B1) was assumed to not generate any emissions. Separate tables are provided for each component, which specify the type of material used, the building assembly it is used in, and the type and quantity of input material consumed (Tables S1-S2). The "Input Unit/Functional Unit" column specifies how much of the input material is consumed for a given unit of each building material. An example of how the table should be interpreted is provided for the first row of Table S1: "5.10E-03 kilograms (kg) of sealant are consumed per kg of aluminum used in a curtain wall system during material maintenance and repair."

<u>Enclosure Component</u>

Table S1. Life cycle inventory inputs for the use phase (B1-B4) of materials used in a building's enclosure, per kilogram (kg) cubic meter (m^3), or square meter (m^2) of material used. The phase column specifies whether the input is relevant to material maintenance and repair (B2-B3) or replacement (B4).

Assembly	Material	Phase	Input Description	Input Quantity	Input Unit/Functional Unit
Curtain wall system	Aluminum	B2-B3	Sealant	5.10E-03	kg/kg
Curtain wall system	Aluminum	B2-B3	Inert waste to landfill	5.10E-04	kg/kg
Curtain wall system	Aluminum	B2-B3	Polyurethane foam	3.82E-04	kg/kg
Windows	Glass	B2-B3	Sealant	1.52E+00	kg/m2
Windows	Glass	B2-B3	Inert waste to landfill	1.52E-01	kg/m2
Windows	Glass	B4	Glass	3.00E+00	m2/m2
Curtain wall system	Glass	B4	Glass waste to landfill	9.77E-02	kg/m2
Curtain wall system	Rubber Gaskets	B4	EPDM gaskets	3.20E-01	kg/kg
Curtain wall system	Rubber Gaskets	B4	Inert waste to landfill	3.36E-01	kg/kg
Curtain wall system	Insulation	B4	Fiberglass insulation	2.00E-01	m2/m2
Curtain wall system	Insulation	B4	Insulation waste to landfill	6.69E-03	kg/kg
Windows	Aluminum Extrusions	B2-B3	Sealant	7.25E-01	kg/kg
Windows	Aluminum Extrusions	B2-B3	Inert waste to landfill	7.25E-02	kg/kg
Windows	Aluminum Extrusions	B4	Aluminum	2.00E+00	kg/kg
Windows	Aluminum Extrusions	B4	Aluminum waste to landfill	1.80E+00	kg/kg
Roofing	TPO Membrane	B2-B3	Electricity	2.22E-01	kg/m2
Roofing	TPO Membrane	B2-B3	TPO membrane	4.29E+00	m2/m2

Roofing	TPO Membrane	B2-B3	Inert waste to landfill	5.92E-01	kg/m2
Roofing	Insulation, Roof	B4	Fiberglass insulation	4.00E+00	m2/m2
Roofing	Insulation, Roof	B4	Insulation waste to landfill	1.27E-01	kg/m2
Roofing	Lumber, Roof	B4	Lumber	4.00E+00	m3/m3
Roofing	Lumber, Roof	B4	Wood to landfill	1.82E+03	kg/m3
Roofing	Plywood, Roof	B4	Plywood	4.00E+00	m3/m3
Roofing	Plywood, Roof	B4	Wood to landfill	2.48E+03	kg/m3
Roofing	Steel, Roof	B4	Galvanized sheet steel	4.00E+00	kg/kg
Roofing	Steel, Roof	B4	Steel waste to landfill	2.80E+00	kg/kg
Roofing	Steel, Roof	B4	Steel waste to recycle	1.20E+00	kg/kg
Roofing	Vapor Barrier	B4	Vapor barrier	4.00E+00	m2/m2
Roofing	Vapor Barrier	B4	Plastic waste to landfill	1.67E+01	kg/m2
Roofing	TPO Membrane	B4	TPO membrane	4.00E+00	m2/m2
Roofing	TPO Membrane	B4	Inert waste to landfill	2.32E+00	kg/m2
Exterior walls	MTP	B2/B3	Stain	7.79E-02	kg/kg
Exterior walls	CLT	B4	Wood to landfill	1.00E+00	kg/kg
Exterior walls	CLT	B4	Nails to landfill	9.03E-04	kg/kg
Exterior walls	CLT	B4	Building paper to landfill	8.29E-02	kg/kg
Exterior walls	CLT	B4	Virgin CLT	2.17E-03	m3/kg
Exterior walls	GLT	B4	Wood to landfill	1.00E+00	kg/kg
Exterior walls	GLT	B4	Nails to landfill	9.03E-04	kg/kg
Exterior walls	GLT	B4	Building paper to landfill	8.29E-02	kg/kg
Exterior walls	GLT	B4	Virgin GLT	1.81E-03	m3/kg
Exterior walls	Concrete	B2/B3	Mortar	2.99E-02	kg/kg
Exterior walls	Concrete	B2/B3	Sealant	2.23E-04	kg/kg
Exterior walls	Concrete	B2/B3	Polyurethane foam	1.68E-05	kg/kg
Exterior walls	Concrete	B4	Concrete to landfill	1.00E+00	kg/kg
Exterior walls	Concrete	B4	Virgin concrete	4.17E-04	m3/kg
Exterior walls	Steel, Cladding	B2/B3	Sealant	2.86E-04	kg/kg
Exterior walls	Steel, Cladding	B4	Virgin steel	1.00E+00	kg/kg
Exterior walls	Steel, Cladding	B4	Steel to recycle	1.65E-01	kg/kg
Exterior walls	Steel, Cladding	B4	Steel to landfill	4.93E-02	kg/kg
Exterior walls	Steel, Cladding	B4	Fasteners to landfill	5.35E-03	kg/kg
Exterior walls	Steel, Cladding	B4	Building paper to landfill	4.91E-01	kg/kg
Exterior walls	Wood, Cladding	B2/B3	Stain	2.16E+00	kg/m2
Exterior walls	Wood, Cladding	B4	Virgin wood	1.00E+00	m2/m2
Exterior walls	Wood, Cladding	B4	Wood to landfill	4.65E+00	kg/m2
Exterior walls	Wood, Cladding	B4	Nails to landfill	2.50E-02	kg/m2
Exterior walls	Wood, Cladding	B4	Building paper to landfill	2.30E+00	kg/m2

Interior Component

Table S2. Life cycle inventory inputs for the use phase (B1-B4) of materials used in a building's enclosure, per kilogram (kg) cubic meter (m^3), or square meter (m^2) of material used. The phase column specifies whether the input is relevant to material maintenance and repair (B2-B3) or replacement (B4).

Assembly	Material	Phase	Input Description	Input Quantity	Input Unit/Functional Unit
Ceiling covering	Acoustic ceiling panels (gypsum)	B4	Inert waste to landfill	1.60E+01	kg/m2
Ceiling covering	Acoustic ceiling panels (gypsum)	B4	Acoustic ceiling panels (gypsum)	2.00E+00	m2/m2
Ceiling covering	Acoustic ceiling panels (mineral wool)	B4	Inert waste to landfill	7.68E+00	kg/m2
Ceiling covering	Acoustic ceiling panels (mineral wool)	B4	Acoustic ceiling panels (mineral wool)	2.00E+00	m2/m2
Finished flooring	Carpet	B4	Inert waste to landfill	4.98E+00	kg/m2
Finished flooring	Carpet	B4	Carpet	7.00E+00	m2/m2
Flooring support	Concrete block	B4	Inert waste to landfill	1.00E+00	kg/kg
Flooring support	Concrete block	B4	Concrete	1.00E+00	kg/kg
Wall covering	Gypsum drywall	B4	Inert waste to landfill	2.15E+01	kg/m2
Wall covering	Gypsum drywall	B4	Gypsum drywall	2.00E+00	m2/m2
Wall covering	Gypsum drywall, type X	B4	Inert waste to landfill	2.15E+01	kg/m2
Wall covering	Gypsum drywall, type X	B4	Gypsum drywall, type X	2.00E+00	m2/m2
Internal wall	Hardwood lumber	B4	Wood waste to landfill	6.10E+02	kg/m3
Internal wall	Hardwood lumber	B4	Hardwood lumber	1.00E+00	m3/m3
Wall covering	Paint	B2/B3	Paint	1.93E+01	kg/kg
Internal wall	Plywood	B4	Wood waste to landfill	6.20E+02	kg/m3
Internal wall	Plywood	B4	Plywood	1.00E+00	m3/m3
Internal wall	Softwood lumber	B4	Wood waste to landfill	4.56E+02	kg/m3
Internal wall	Softwood lumber	B4	Softwood lumber	1.00E+00	m3/m3
Internal wall supports	Steel, galvanized	B4	Steel waste to recycle	7.70E-01	kg/kg
Internal wall supports	Steel, galvanized	B4	Inert waste to landfill	2.30E-01	kg/kg
Internal wall supports	Steel, galvanized	B4	Steel, galvanized	1.00E+00	kg/kg
Internal wall supports	Steel, hollow structural sections	B4	Steel waste to recycle	7.70E-01	kg/kg

Internal wall supports	Steel, hollow structural sections	B4	Inert waste to landfill	2.30E-01	kg/kg
Internal wall supports	Steel, hollow structural sections	B4	Steel, hollow structural sections	1.00E+00	kg/kg
Internal wall	Steel, partition wall	B4	Steel waste to recycle	9.61E-02	kg/m2
Internal wall	Steel, partition wall	B4	Inert waste to landfill	2.87E-02	kg/m2
Internal wall	Steel, partition wall	B4	Steel, partition wall	1.00E+00	m2/m2
Internal wall supports	Steel, sheet (studs)	B4	Steel waste to recycle	7.70E-01	kg/kg
Internal wall supports	Steel, sheet (studs)	B4	Inert waste to landfill	2.30E-01	kg/kg
Internal wall supports	Steel, sheet (studs)	B4	Steel, sheet	1.00E+00	kg/kg
Internal wall supports	Structural steel	B4	Steel waste to recycle	7.70E-01	kg/kg
Internal wall supports	Structural steel	B4	Inert waste to landfill	2.30E-01	kg/kg
Internal wall supports	Structural steel	B4	Structural steel	1.00E+00	kg/kg
Finished flooring	Tile	B4	Inert waste to landfill	4.08E+01	kg/m2
Finished flooring	Tile	B4	Tile	2.00E+00	m2/m2
Wall covering	Tile	B4	Inert waste to landfill	4.08E+01	kg/m2

Appendix B

This appendix contains the survey questions used in Chapter 2 of this dissertation. In this chapter, two surveys were conducted. The first survey ("Waste Transportation") was conducted for the waste transportation analysis, in which certified C&D waste processing facilities in Sunnyvale, CA were surveyed to determine the average distance traveled by different types of C&D waste during waste transportation. The second survey ("Metal Recycling") was conducted for the waste processing analysis, in which metal scrap yards and recyclers in King County, Washington were surveyed to determine the amounts and types of energy and materials consumed during steel and aluminum recycling.

Waste Transportation Survey

1. Is your facility classified as an MRF, transfer station, reuse facility, or recycler?
2. Do you accept CLT or glulam?
3. For the following types of materials, where do you send outgoing waste to be recycled? If you are not able to provide the name of the facility due to confidentiality reasons, the name of the city or an estimate of the distance suffice:
    - C&D wood
    - C&D concrete
    - C&D steel
    - C&D drywall/gypsum
4. Where do you send C&D waste that needs to be landfilled?
5. Is your outbound waste shipped by truck or train?

Metal Recycling Survey

1. Can you describe the scrap treatment process for all incoming scrap? (e.g., sorting, washing, purification)
2. What types of machines or equipment are used for each step? If possible, please include the manufacturer and model.
3. How many hours does each machine run daily?
4. How much scrap do you process daily?

Appendix C

This appendix contains the life cycle inventory (LCI) data for waste processing 1 kilogram of different construction and demolition (C&D) materials in Chapter 2 of this dissertation. The material and energy inputs required for the waste processing (C3) life cycle module are reported in separate tables for each material, on a one kilogram basis. Data are provided for concrete, reinforced concrete, rebar, steel, aluminum, and inert waste (Tables S3-S8).

Unlike the other C&D materials in this appendix, reinforced concrete is a composite material that enters the MRF as one material and leaves as two separate materials: concrete aggregate and rebar. Analyzing the proportion of concrete to rebar in reinforced concrete in the two case study buildings revealed that the concrete-to-rebar ratio was higher in the structure than in the foundation; however, more reinforced concrete was used in the structure than the foundation. This resulted in a weighted average concrete-to-rebar ratio of 0.94 kg of concrete to 0.06 kg of rebar in both buildings. Thus, the LCI in Table S4 is allocated by mass according to this ratio of products.

Concrete

Table S3. Life cycle inventory inputs for waste processing 1 kilogram of waste concrete at a material recovery facility (MRF).

Input (per kg concrete)	Quantity	Unit	Process Description
MRF			
Electricity	0.001859	kWh	Facility operations
Propane (RFO)	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.000617	kWh	Conveyer (picking line)
Electricity	0.007385	kWh	Rock crusher
Electricity	0.021129	kWh	Magnet (pre-crushing and during crushing)

Reinforced Concrete

Table S4. Life cycle inventory inputs for waste processing 1 kilogram of waste reinforced concrete, unallocated.

C3- Waste Processing	Quantity	Unit	Process Description
Outputs			
Concrete aggregate	0.94	kg	
Rebar	0.06	kg	
Inputs			
Electricity	0.001859	kWh	Facility operations
Propane (RFO)	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.000617	kWh	Conveyer (picking line)
Electricity	0.007385	kWh	Rock crusher
Electricity	0.021129	kWh	Magnet (pre-crushing and during crushing)

Rebar

Table S4. Life cycle inventory inputs for waste processing 1 kilogram of waste rebar at a material recovery facility (MRF) and a scrap yard.

Input (per kg rebar)	Quantity	Unit	Process Description
MRF			
Electricity	0.001859	kWh	Facility operations
Propane (RFO)	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.000617	kWh	Conveyer (picking line)
Electricity	0.007385	kWh	Rock crusher
Electricity	0.021129	kWh	Magnet (pre-crushing and during crushing)
Scrap Yard			
Diesel	4.24E-05	L	Front-end loader, sorting
Diesel	7.80E-05	L	Backhoe, sorting
Diesel	7.36E-05	L	Sennebogen, sorting
Diesel	2.21E-02	L	Backhoe, shearing
Electricity	4.24E-05	kWh	Industrial shredder, shredding

Steel

Table S5. Life cycle inventory inputs for waste processing 1 kilogram of waste steel at a material recovery facility (MRF) and a scrap yard.

Input (per kg steel)	Quantity	Unit	Process Description
MRF			
Electricity	0.001859	kWh	Facility operations
Propane (RFO)	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.010565	kWh	Magnet
Scrap Yard			
Diesel	4.24E-05	L	Front-end loader, sorting
Diesel	7.80E-05	L	Backhoe, sorting
Diesel	7.36E-05	L	Sennebogen, sorting
Diesel	2.21E-02	L	Backhoe, shearing
Electricity	4.24E-05	kWh	Industrial shredder, shredding

Aluminum

Table S6. Life cycle inventory inputs for waste processing 1 kilogram of waste aluminum at a material recovery facility (MRF) and a scrap yard.

Input (per kg aluminum)	Quantity	Unit	Process Description
MRF			
Electricity	0.000404	kWh	Facility operations
Propane	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.021129	kWh	Magnet
Electricity	0.000617	kWh	Conveyer (picking line)
Scrap Yard			
Diesel	1.16E-04	L	Sorting & compressing
Diesel	7.80E-05	L	Shearing

Gypsum

Table S7. Life cycle inventory inputs for waste processing 1 kilogram of waste gypsum at a material recovery facility (MRF) and a recycling plant.

Input (per kg gypsum)	Quantity	Unit	Process Description
MRF			
Electricity	0.000404	kWh	Facility operations
Propane (LPG)	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.010565	kWh	Magnet
Electricity	0.000617	kWh	Conveyer (picking line)
Recycling Plant			
Electricity	0.03261	kWh	Crushing, in impact mill
Diesel	0.0009	L	Separation
Electricity	0.01048	kWh	Milling
Electricity	0.01123	kWh	Packaging & storage
Electricity	0.00603	kWh	Internal transport

Inert Waste

Table S8. Life cycle inventory inputs for waste processing 1 kilogram of inert waste at a material recovery facility (MRF).

Input (per kg inert)	Quantity	Unit	Process Description
MRF			
Electricity	0.000404	kWh	Facility operations
Propane (RFO)	0.000867	L	Rolling stock
Electricity	0.003489	kWh	Trommel
Electricity	0.010565	kWh	Conveyer (picking line)
Electricity	0.000617	kWh	Rock crusher
Electricity	0.000404	kWh	Magnet (pre-crushing and during crushing)

Appendix D

This appendix contains the end-of-life (EOL) cycle impact assessment (LCIA) results for 1 kilogram of different construction and demolition (C&D) materials in Chapter 2 of this dissertation. The LCIA results are reported by life cycle module, including waste transportation to processing facilities (C2), waste processing (C3), and waste disposal (C4). Building demolition or deconstruction (C1) are excluded,

The EOL phase LCIA results are reported for each material under all applicable EOL scenarios (Table S9). The LCIA results for incinerating mass timber panels (MTP) differ between the two types of MTP considered, so the impacts of incineration are reported separately for cross-laminated timber (CLT) and glue-laminated timber (GLT).

Table S9. End-of-life (EOL) phase (C2-C4) impacts per kilogram of construction and demolition (C&) materials in different EOL scenarios.

Material	EOL Scenario	GWP_{FOSSIL} (kg CO_2e)	AP (kg SO_2e)	EP (kg Ne)	ODP (kg CFC_{-11}e)	SFP (kg O_3e)
Aluminum	Recycle	9.65E-03	3.27E-05	3.71E-05	8.09E-11	5.70E-04
Aluminum	Landfill	3.75E-02	2.40E-04	7.45E-05	5.32E-09	3.98E-03
Concrete	Recycle	1.33E-02	3.63E-05	5.74E-05	1.25E-10	2.45E-04
Concrete	Landfill	2.67E-02	1.28E-04	8.40E-05	4.21E-09	2.77E-03
Gypsum	Recycle	4.08E-02	1.44E-04	1.83E-04	3.91E-10	2.38E-03
Gypsum	Landfill	2.80E-02	3.09E-02	5.88E-05	5.32E-09	3.61E-03
Inert	Landfill	2.24E-02	1.16E-04	6.30E-05	4.16E-09	2.82E-03
MTP	Reprocess	1.76E-01	2.15E-03	4.88E-04	1.06E-09	3.47E-02
MTP	Reuse	8.86E-02	9.34E-04	8.97E-05	2.09E-10	2.94E-02
MTP	Recycle	2.80E-02	2.26E-04	5.59E-05	1.25E-10	6.68E-03
CLT	Incinerate	8.80E-02	3.15E-04	7.48E-04	8.93E-10	9.92E-03
GLT	Incinerate	9.04E-02	3.17E-04	7.50E-04	8.96E-10	9.97E-03
MTP	Landfill	1.68E-01	1.19E-04	7.16E-03	4.16E-09	2.84E-03
Rebar	Recycle	2.95E-02	8.84E-05	1.36E-04	2.91E-10	1.15E-03
Rebar	Landfill	3.17E-02	1.42E-04	1.09E-04	4.26E-09	3.09E-03
Reinforced concrete	Recycle	1.83E-02	5.05E-05	8.25E-05	1.78E-10	5.71E-04
Reinforced concrete	Landfill	3.17E-02	1.42E-04	1.09E-04	4.26E-09	3.09E-03
Steel	Recycle	1.99E-02	6.15E-05	8.81E-05	1.90E-10	8.68E-04
Steel	Landfill	2.21E-02	1.15E-04	6.16E-05	4.16E-09	2.81E-03
Wood, non-MTP	Recycle	2.80E-02	2.26E-04	5.59E-05	1.25E-10	6.68E-03
Wood, non-MTP	Incinerate	4.07E-02	5.17E-04	7.86E-04	9.68E-10	1.59E-02
Wood, non-MTP	Landfill	1.70E-01	1.20E-04	7.27E-03	4.18E-09	2.86E-03

Appendix E

This appendix contains the life cycle impact assessment (LCIA) results for the MT1 and Concrete Baseline case study buildings assessed in Chapter 4 of this dissertation. Results are reported by life cycle phase, in different waste scenarios and carbon storage accounting methods:

- **Business-as-Usual (BAU):** Demolition + BAU disposal
  - BAU ISO
  - BAU Radiative Forcing
- **Optimistic:** Selective deconstruction + reuse of MTP
  - Optimistic ISO
  - Optimistic Radiative Forcing

All four scenarios were assessed for the MT1 building; only the BAU ISO scenario was assessed for the Concrete Baseline. The LCIA results are reported by life cycle phase, including production/construction (A1-A5), use (B1-B4), end-of-life (C1-C4), and virgin product substitution benefits (Module D). The carbon storage benefits from biogenic carbon in wood and carbonation in concrete are also evaluated.

The LCIA results are reported by life cycle phase for each building under all applicable waste scenarios (Table S10).

Table S10. Life cycle impact assessment results for each case study building under different waste scenarios, by life cycle phase.

Building	Waste Scenario	Life Cycle Phase	GWP_{FOSSIL} (kg CO_2e)	AP (kg SO_2e)	EP (kg Ne)	ODP (kg $CFC_{-11}e$)	SFP (kg O_3e)
MT1	BAU ISO	A1-A5	7.28E+06	3.46E+04	7.65E+03	1.96E-01	5.88E+05
MT1	BAU ISO	B2-B4	6.40E+06	2.05E+04	5.28E+03	3.10E-01	3.26E+05
MT1	BAU ISO	C1-C4	8.73E+05	4.34E+03	1.76E+04	2.52E-02	1.16E+05
MT1	BAU ISO	D	-2.24E+06	-7.79E+03	-9.12E+03	-2.43E-02	-9.88E+04
MT1	BAU ISO	Carbon	-3.96E+06	0.00E+00	0.00E+00	0.00E+00	0.00E+00
MT1	BAU Avoided Emissions	A1-A5	7.28E+06	3.46E+04	7.65E+03	1.96E-01	5.88E+05
MT1	BAU Avoided Emissions	B2-B4	6.40E+06	2.05E+04	5.28E+03	3.10E-01	3.26E+05
MT1	BAU Avoided Emissions	C1-C4	8.73E+05	4.34E+03	1.76E+04	2.52E-02	1.16E+05
MT1	BAU Avoided Emissions	D	-2.24E+06	-7.79E+03	-9.12E+03	-2.43E-02	-9.88E+04

MT1	BAU Avoided Emissions	Carbon	-5.54E+06	0.00E+00	0.00E+00	0.00E+00	0.00E+00
MT1	Optimistic ISO	A1-A5	7.28E+06	3.46E+04	7.65E+03	1.96E-01	5.88E+05
MT1	Optimistic ISO	B2-B4	6.36E+06	2.02E+04	5.13E+03	3.05E-01	3.24E+05
MT1	Optimistic ISO	C1-C4	7.03E+05	6.47E+03	2.04E+03	1.63E-02	1.85E+05
MT1	Optimistic ISO	D	-2.75E+06	-1.22E+04	-9.38E+03	-9.33E-02	-2.56E+05
MT1	Optimistic ISO	Carbon	-4.37E+05	0.00E+00	0.00E+00	0.00E+00	0.00E+00
MT1	Optimistic Avoided Emissions	A1-A5	7.28E+06	3.46E+04	7.65E+03	1.96E-01	5.88E+05
MT1	Optimistic Avoided Emissions	B2-B4	6.36E+06	2.02E+04	5.13E+03	3.05E-01	3.24E+05
MT1	Optimistic Avoided Emissions	C1-C4	7.03E+05	6.47E+03	2.04E+03	1.63E-02	1.85E+05
MT1	Optimistic Avoided Emissions	D	-2.75E+06	-1.22E+04	-9.38E+03	-9.33E-02	-2.56E+05
MT1	Optimistic Avoided Emissions	Carbon	-5.87E+06	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Concrete Baseline	BAU ISO	A1-A5	1.06E+07	3.76E+04	8.79E+03	2.62E-01	7.39E+05
Concrete Baseline	BAU ISO	B2-B4	8.00E+06	3.26E+04	8.36E+03	4.68E-01	6.04E+05
Concrete Baseline	BAU ISO	C1-C4	5.13E+05	3.64E+03	2.07E+03	2.31E-02	8.99E+04
Concrete Baseline	BAU ISO	D	-3.17E+06	-1.11E+04	-1.33E+04	-3.55E-02	-1.47E+05
Concrete Baseline	BAU ISO	Carbon	-9.85E+05	0.00E+00	0.00E+00	0.00E+00	0.00E+00