Improving Organizational Life Cycle Assessment (O-LCA) through a Hospital Case Study

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

I am the sole author of Chapter 1 and Chapter 5 of this thesis. Chapters 2-4 are based on works that are co-authored with other contributors. Chapter 2 is based on an article published in the *Journal of Industrial Ecology*, on which I am the lead author, joined by co-authors Jair Santillán-Saldivar, Cassandra Thiel, Guido Sonnemann, and Steven B. Young. Chapter 3 is based on a manuscript under review in the *Journal of Industrial Ecology*, on which I am the lead author, joined by co-author Steven B. Young. Chapter 4 is based on a manuscript under review in the *Journal of Industrial Ecology*, on which I am the lead author, joined by co-author Steven B. Young. Bibliographic details for these co-authored works are provided below:

Chapter 2: Potential for industrial ecology to support healthcare sustainability: Scoping review of a fragmented literature and conceptual framework for future research

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Chapter 3: Improving organizational LCA

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Chapter 4: Organizational LCA of a hospital

Cimprich, A., & Young, S.B. *Journal of Industrial Ecology* (in review; manuscript no. 22-JIE-7374)

Abstract

In this thesis, I advance methods and data for organizational life cycle assessment (O-LCA) through a novel application that supports sustainability management in healthcare. In essence, the term *sustainability management* broadly encompasses decision-making for sustainability – considering environmental, social, and economic aspects – at all levels of coupled human and natural systems (*e.g.*, at the level of individuals, organizations, municipalities, provinces, countries, and international bodies). In Chapter 1, I establish the conceptual foundations for the work presented in Chapters 2-4. In Chapter 2 (co-authored with Jair Santillán-Saldivar, Cassandra Thiel, Guido Sonnemann, and Steven B. Young), I conduct a comprehensive scoping review of the literature on "healthcare sustainability" – based on a representative sample from over 1,700 articles published between 1987 and 2017 – that highlights largely untapped opportunities to use *industrial ecology* approaches (such as LCA) to build an evidence base for this burgeoning domain of sustainability management. In Chapter 3 (co-authored with Steven B. Young), I constructively critique the existing methodology for O-LCA, with concrete proposals – particularly the use of basic statistical sampling and inference techniques – to strike a better balance of scientific rigour and practical feasibility in O-LCA. In Chapter 4 (co-authored with Steven B. Young), I test and demonstrate these proposals through an O-LCA of a Canadian hospital, in which I compiled new LCA data for approximately 200 goods and services used in healthcare. Finally, in Chapter 5, I reflect upon my contributions in this thesis, and on opportunities for future work.

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I would also like to thank Cassandra Thiel, Guido Sonnemann, and Steven B. Young, along with Jair Santillán-Saldivar, as co-authors on Chapter 2 of this thesis, which is also published in the *Journal of Industrial Ecology* (Cimprich et al., 2019).

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List of Abbreviations

Life Cycle Assessment	LCA
Organizational Life Cycle Assessment	O-LCA
Life Cycle Inventory	LCI
Life Cycle Impact Assessment	LCIA
Environmentally Extended Input-Output Life Cycle Assessment	EEIO-LCA
Tool for the Reduction and Assessment of Chemical and other environmental Impacts	TRACI
Greenhouse Gas	GHG
Global Warming Potential	GWP

Chapter 1

1. Introduction

In this thesis, I advance methods and data for organizational life cycle assessment (O-LCA) through a novel application that supports sustainability management in healthcare. In this chapter, I establish the conceptual foundations of the thesis. I begin by broadly conceptualizing *sustainability management* – as the namesake of my doctoral degree program. Within this realm, I turn to the core principles and concepts of *industrial ecology*, focusing on the general methodological framework of LCA. Finally, I outline the work presented in Chapters 2-4.

1.1 Sustainability management

Given that it is the namesake of my doctoral degree program, I think it is important to begin by articulating – from an academic and a practical perspective – what "sustainability management" is. To state the obvious, the term combines two key words: *sustainability* and *management*. In this section, I briefly outline the semantics of these terms – first in isolation, and then in combination – for the purpose of my degree program.

The Cambridge Dictionary defines the verb *sustain* as "to cause or allow something to continue for a period of time" (Cambridge University Press, 2021b). The basic notion of sustainability has a myriad of contextual applications (*e.g.*, sustainable growth [of X], financial sustainability, sustainable competitive advantage, *etc.*). The *Sustainability Management* program, however, concerns a broad concept commonly known as *sustainable development*.

The term “sustainable development” was coined by the World Commission on Environment and Development in its landmark report entitled *Our Common Future* (WCED, 1987). As the WCED was chaired by Gro Harlem Brundtland – then Prime Minister of Norway – the report is commonly referred to as “the Brundtland report.” The report contains the following text, which articulates what is probably the most widely cited definition of sustainable development:

> …humanity has the ability to make development sustainable – to ensure that it meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987, p. 8).

As Kates et al. (2005) put it, the notion of sustainable development served as a “grand compromise” between the competing objectives of environmental protection and economic development. Embedded in the Brundtland definition are the core concepts of “needs” (particularly the needs of the world’s poor), “limits” (particularly limits on the capacity of the natural environment to provide resources and assimilate wastes from human activities), and equity (both between present and future generations – commonly termed *inter*generational equity, and within a given generation – commonly termed *intra*generational equity).

Following the publication of the Brundtland report, another highly influential conceptualization of sustainable development (or just “sustainability,” as the terms are often used interchangeably) is the notion of the “triple bottom line” – coined by John Elkington, co-founder of the business management consultancy named SustainAbility, in his seminal book entitled *Cannibals with Forks: The Triple Bottom Line of 21st Century Business* (Elkington, 1998). In his book, Elkington uses the metaphor of a three-pronged fork representing the “triple bottom line” of sustainability – comprising economic prosperity, environmental quality, and social justice.

Though *Cannibals with Forks* was written primarily through a business management lens, the essence of the triple bottom line – also commonly known as the "three pillars of sustainability" (along with other variations, such as "people, planet, profit") – has permeated the sustainability discourse, both within and outside of academia (see, *e.g.*, Gibson (2006), Isil & Hernke (2017), Miller (2020), and The Economist (2009)). Elkington adjusted the Brundtland definition of sustainable development to incorporate the triple bottom line concept:

> Sustainability is the principle of ensuring that our actions today do not limit the range of economic, social, and environmental options open to future generations (Elkington, 1998, p. 20).

Around the same time as the publication of *Cannibals with Forks*, the U.S. National Academy of Sciences published its report, entitled *Our Common Journey: A Transition Toward Sustainability,* in which the concept of sustainable development was deconstructed into a set of four simple – yet challenging – questions (Kates et al., 2005; U.S. National Academy of Sciences, 1999):

1. What is to be sustained?
2. What is to be developed?
3. What are the links between what is to be sustained and what is to be developed?
4. What is the time horizon?

In response to the first question – "what is to be sustained" – three broad categories were highlighted: "nature" (comprising ecosystems, biodiversity, and the Earth), "life support" systems (comprising the environment, resources, and ecosystem services), and "community" (comprising places, groups, and cultures) (Kates et al., 2005; U.S. National Academy of

Sciences, 1999). On the second question – “what is to be developed” – the highlighted categories were: “people” (comprising child survival, life expectancy, education, equity, and equal opportunity), “economy” (comprising wealth, productive sectors, and consumption), and “society” (comprising institutions, social capital, states, and regions) (Kates et al., 2005; U.S. National Academy of Sciences, 1999). Responses to the third question – regarding the links between what is to be sustained and what is to be developed – ranged from “sustain only,” to “develop only,” to various forms of “and/or” (Kates et al., 2005; U.S. National Academy of Sciences, 1999). Finally, responses to the fourth question – regarding the time horizon – ranged from a single generation (appx. 25 years), to several generations, to “forever” (Kates et al., 2005; U.S. National Academy of Sciences, 1999).

More recently, with the environmental stresses from human activities approaching, if not already crossing, several “planetary boundaries” (Steffen et al., 2015), and with millions of people continuing to experience living standards that fall below the “social foundation” (Raworth, 2012), the United Nations adopted a set of 17 *Sustainable Development Goals* (SDGs) – building upon the *Millennium Development Goals* (MDGs) – as part of *The 2030 Agenda for Sustainable Development* (*i.e.*, “Agenda 2030”):

1. End poverty in all its forms everywhere
2. End hunger, achieve food security and improved nutrition and promote sustainable agriculture
3. Ensure healthy lives and promote well-being for all at all ages
4. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all
5. Achieve gender equality and empower all women and girls

6. Ensure availability and sustainable management of water and sanitation for all
7. Ensure access to affordable, reliable, sustainable and modern energy for all
8. Promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all
9. Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation
10. Reduce inequality within and among countries
11. Make cities and human settlements inclusive, safe, resilient and sustainable
12. Ensure sustainable consumption and production patterns
13. Take urgent action to combat climate change and its impacts
14. Conserve and sustainably use the oceans, seas and marine resources for sustainable development
15. Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss
16. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable and inclusive institutions at all levels
17. Strengthen the means of implementation and revitalize the Global Partnership for Sustainable Development (United Nations, 2015)

Kates et al. (2005) describe the concept of sustainable development as “creatively ambiguous.” That strikes me as quite a diplomatic way of putting it; I would more bluntly describe the notion of sustainable development (or “sustainability”) as *nebulous*. As can be seen from the selected foundational texts cited above, the concept is extremely broad and difficult to operationalize. Nonetheless, three core features are evident:

1. Sustainability is primarily a *normative* concept (which creates tensions between science, policy, and practice);
2. It is a *forward-looking* (or *future-oriented*) concept, with a strong sense of urgency; and
3. It is a *multifaceted* (or, from a scholarly perspective, *multidisciplinary*) concept (*e.g.*, having complex and interrelated environmental, economic, and social dimensions).

What is sustainability *management*, then? For many people, the first thing that comes to mind may be the integration of sustainability in *business* management. Indeed, the School of Environment, Enterprise and Development offers academic programs and courses – at the undergraduate and graduate level – with the title of “Environment and Business” (though arguably “*Sustainability* and Business” may be more accurate). While “business” does play an important role in sustainability management (see, *e.g.*, the first thesis accepted in the Sustainability Management doctoral program – with a focus on *corporate social responsibility* – by ElAlfy (2020)), the basic concept is much broader. The Cambridge Dictionary defines the verb “manage,” depending on the context, as:

> to succeed in doing or dealing with something, especially something difficult […];
>
> to control or organize someone or something […]; or
>
> to be able to use something, for example time or money, in an effective way (Cambridge University Press, 2021a).

Thus, broadly speaking, sustainability management is about using "things" (*e.g.*, time, money, and knowledge) to control and organize efforts to achieve sustainability objectives. Perhaps the most explicit and comprehensive definition of sustainability management is formulated in a widely-cited (with 179 citations, according to the Scopus database, as of July 8, 2021) theoretical article by Starik & Kanashiro (2013):

> We define sustainability management as the formulation, implementation, and evaluation of both environmental and socioeconomic sustainability-related *decisions and actions* […] and, for the purposes of this article, includes *decisions and actions* at the individual, organizational, and societal levels (p. 12, emphasis added).

In essence, the term *sustainability management* broadly encompasses decision-making for sustainability – considering environmental, social, and economic aspects – at all levels of coupled human and natural systems (*e.g.*, at the level of individuals, organizations, municipalities, provinces, countries, and international bodies). The scholarly discourse on sustainability management is scattered across a wide variety of disparate bodies of literature, with contributions from the natural sciences, social sciences, humanities, and health sciences. As of July 8, 2021, the Scopus source title list (available for download from Elsevier) contains nearly 600 journals with the words "environment*" and/or "sustainab*" in the title (the asterisk is used to capture all possible suffixes). Sustainability management literature can also be found in "mainstream" outlets like the *Academy of Management Journal* (*e.g.*, Russo & Fouts (1997); Wright & Nyberg (2017)), the *Academy of Management Review* (*e.g.*, Donaldson & Preston (1995); Gladwin et al. (1995)), the *Harvard Business Review* (*e.g.*, Lovins et al. (2007); Porter &

Kramer (2006, 2011)), *Science* (*e.g.*, Burke et al. (2016); Daily et al. (2000); Dietz et al. (2003); Hsiang et al. (2017); Steffen et al. (2015)), and the *Proceedings of the National Academy of Sciences (PNAS)* (*e.g.*, Graedel et al. (2015)). Beyond the nebulous concept of "sustainable development," it is difficult to discern an overarching framework or theoretical orientation that ties this fragmented literature together.

Within the broader realm of sustainability management, this thesis is grounded in the conceptual foundations of *industrial ecology* – itself a broad and interdisciplinary field, as briefly described in the next section.

1.2 Industrial ecology

To state the obvious again, the term "industrial ecology" combines the words *industrial* and *ecology*. The word "ecology" is derived from the Greek words *eco* (meaning "home") and *logy* (meaning "study"); in general, any term ending with the suffix "…*ology*" refers to the *study of* something. *Ecology* can be defined as "the study of the home" (or, more completely, "the study of organisms in their home"). The relatively new field of *industrial ecology*, emerging from the idea of "industrial ecosystems" (commonly credited to Frosch & Gallopoulos (1989), in what may be considered one of the founding contributions to the field of industrial ecology), essentially extends the conceptual foundations of ecology – which is traditionally concerned with "natural" ecosystems – to study the "industrial" ecosystems of human civilizations. The word "industrial" is derived from the Latin word *industria*, which roughly translates to "activity" – particularly *human* activity. Accordingly, industrial ecology can be broadly defined as the study of the activities of humans in their “home,” where the “home” could be defined at any scale – from a facility or site, to a country or region, to the entire planet Earth (and beyond?).

In his preface to the seminal book entitled *The Greening of Industrial Ecosystems* (1994), Robert White – then President of the U.S. National Academy of Engineering – defined industrial ecology as:

> …the study of the flow of materials and energy in industrial and consumer activities, of the effects of these flows on the environment, and of the influences of economic, political, regulatory, and social factors on the flow, use, and transformation of resources

In other words, industrial ecology is a broad, interdisciplinary field of study that blurs the lines between the natural sciences, engineering, social sciences, and to some extent even health sciences (*e.g.*, as seen in the inclusion of *human health* as an “area of protection” in the environmental “life cycle assessment” (LCA) of goods and services (Hauschild et al., 2013)). It is essentially the study of matter and energy flows – and the environmental, economic, and social impacts (*i.e.*, “sustainability”) of these flows – associated with human activities throughout the “life cycle” of goods and services in the economy – from the “cradle” where resources are extracted from the environment to the “grave” where wastes are ultimately returned to the environment (see, *e.g.*, the book by Graedel & Allenby (2003), which is the second edition of what may be considered the first textbook on industrial ecology). Like the “natural” ecology (for lack of better term) from which it is inspired, industrial ecology follows fundamental principles – like the laws of thermodynamics and the law of conservation of mass – from the natural sciences and engineering (Graedel & Allenby, 2003; Kleijn, 2000). And, like natural ecology, industrial ecology is grounded in *systems thinking*.

The Cambridge Dictionary defines a *system* as “a set of connected things or devices that operate together” or “a way of doing things” (Cambridge University Press, 2022); it is the former notion that is most relevant here. The term “general systems theory” is widely credited to Austrian biologist Ludwig Von Bertalanffy, who, after quoting Aristotle’s declaration that “the whole is more than the sum of its parts,” defined a *system* as “…a set of elements standing in interrelation among themselves and with the environment” (Von Bertalanffy, 1972, p. 417). This definition aptly describes the kinds of systems studied in both natural ecology and industrial ecology. Natural ecologists study natural ecosystems – sets of “elements” (*i.e.*, organisms) standing in relation among themselves and with the environment (*i.e.*, abiotic elements like air, water, and land). Industrial ecologists study industrial ecosystems, which also comprise sets of “elements” (*e.g.*, materials, chemicals, products, people, and organizations) standing in relation among themselves and with the environment (both “natural” and “human-made”). A system can itself be a part of a larger system (*e.g.*, an organism is itself a system, which in turn comprises many subsystems, and a product, such as an automobile, can similarly be described as a “system of systems”).

Delimiting a *system boundary* (*i.e.*, to limit the scope of the “systems of systems” studied) – however artificial it may be – is fundamentally necessary in both natural and industrial ecology. System boundary definition – or *goal and scope definition*, to use the terminology in the international standards (ISO 14040:2006, ISO 14044:2006) on life cycle assessment (LCA) – is central to all methodological approaches of industrial ecology, including LCA, material flow analysis (MFA), and economic input-output analysis. In the next section, I focus on the methodological framework of LCA – as the foundation for the work in this thesis.

1.3 Life cycle assessment

LCA is a widely used and internationally standardized approach, facilitated through commercially available and open-source software and extensive databases, for systematically evaluating the environmental impacts of a *product system* (wherein the definition of “product” includes any good or service). More precisely, the International Organization for Standardization (ISO) defines LCA as “[the] compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle,” wherein the *life cycle* is defined as “consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal” (ISO 14040:2006, ISO 14044:2006). As illustrated in Figure 1, the LCA framework comprises four interconnected methodological “phases”: goal and scope definition, life cycle inventory (LCI) analysis, life cycle impact assessment (LCIA), and interpretation (ISO 14040:2006, ISO 14044:2006).

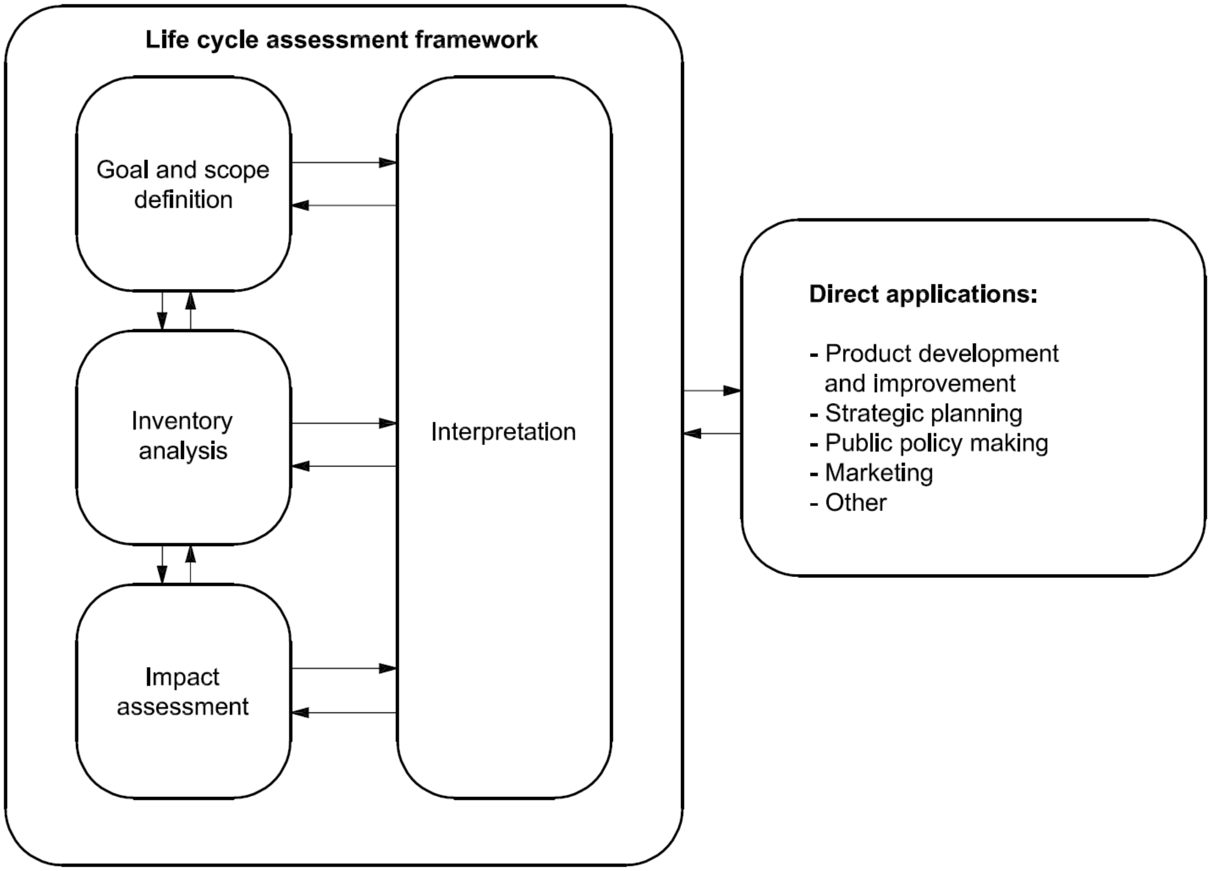


Figure 1: Life cycle assessment (LCA) framework (ISO 14040:2006, Figure 1)

The goal of an LCA needs to be defined in terms of the intended application(s) of the LCA, the reason(s) for carrying out the LCA, the intended audience(s) to whom the LCA results will be communicated, and whether the results are intended to be used in comparative assertions (*i.e.*, "product A is better for the environment than product B") intended to be disclosed to the public (ISO 14044:2006). The scope needs to be defined, among other things, in terms of the product system studied (or systems, if the LCA is comparative), the function(s) of the product system(s), the "functional unit" of the product system(s), and the system boundary (ISO 14044:2006).

The notion of a *functional unit* is central to LCA goal and scope definition. ISO 14044 defines the functional unit as "quantified performance of a product system for use as a reference unit" (sec. 3.20). It continues: "One of the primary purposes of a functional unit is to provide a reference to which the input and output data are normalized (in a mathematical sense). Therefore the functional unit shall be clearly defined and measurable" (sec. 4.2.3.2). In other words, the functional unit quantifies the core use or purpose of a product. It defines the object of study – the actual "product" studied in the LCA – and, in the case of a comparative LCA, forms the basis for an "apples-to-apples" comparison. Consider, *e.g.*, a comparative LCA of a hand dryer versus paper towels. Ultimately, the "product" of interest is not the hand dryer or the paper towels, but rather *the function of drying hands*. Accordingly, the *functional unit* could be defined in terms of pairs of hands dried. Put another way, the functional unit is the "end," and the product system(s) is(are) the "means." Comparative LCAs effectively compare the environmental impacts of alternative *means* (*i.e.*, product systems) of achieving a common *end* (*i.e.*, a functional unit).

Equally important is the *system boundary* definition, which delimits the "unit processes" (*i.e.*, specific processes for resource extraction, materials and chemicals production, product manufacturing, product use, and product "end-of-life" management) included within the LCA and is typically illustrated using a process flow diagram. In practice, it is typically not feasible for the system boundary to encompass *every* unit process, especially for complex high-tech products like, *e.g.*, an automobile. Therefore, ISO 14044 allows for "cut-off" criteria (*i.e.*, based on mass or energy contribution, or based on presumed environmental significance) to exclude some parts of the product system that are deemed to be of comparatively limited significance. "Cut-offs" can be seen as a weakness of LCA, which can be addressed by combining "process-based" LCA (*i.e.*, per ISO 14040 and 14044) with environmentally extended input-output

(EEIO)-LCA – an approach that combines economic input-output analysis (a methodology, developed by Nobel laureate Wassily Leontief, that accounts for sector-level transactions in aggregated monetary values) with information on the emissions intensity of economic sectors (see, *e.g.*, the pioneering work by Hendrickson et al. (1998)). This combination of process-based LCA and EEIO-LCA is widely known as "hybrid" LCA.

Following the goal and scope definition, life cycle inventory (LCI) analysis is "[the] phase of [LCA] involving the compilation and quantification of inputs and outputs for a product throughout its life cycle" (ISO 14044:2006, sec. 3.3). In the LCI phase, data are collected to describe and quantify the inputs and outputs of each unit process in the defined system boundary. When the system boundary is defined properly in conformance with ISO 14044, all inputs and outputs crossing the system boundary will be in the form of *elementary flows* – defined as "material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation" (sec. 3.12). In other words, the result of LCI analysis is essentially a list of all the elementary flows – all the inputs (*i.e.*, withdrawals from air, water, and soil) and outputs (*i.e.*, emissions to air, water, and soil) that cross the boundary between the product system (sometimes referred to as the "technosphere") and the environment (sometimes referred to as the "ecosphere").

In practice, LCI analysis is facilitated using extensive databases – like those maintained by the Swiss organization Ecoinvent – embedded in LCA software programs like SimaPro and OpenLCA (see links in reference list). These databases provide what is commonly referred to as "background data" in LCA practice (see, *e.g.*, Ecoinvent (2021), European Commission (2010), Life Cycle Initiative (2021), and US EPA (2006)) – *i.e.*, data on the production of widely used goods and services (*e.g.*, materials, chemicals, fuels, electricity, water, and waste treatment). These background data are combined with "foreground data" specific to the studied product system (*e.g.*, material composition, manufacturing processes, energy and water use, and waste treatment routes) to derive the LCI results for the product system.

Using LCA software and databases, LCI analysis typically yields a list of *thousands* of elementary flows; this information is difficult to interpret, and quite meaningless to decision-makers (Hauschild & Huijbregts, 2015). Moreover, different elementary flows are incommensurable (*e.g.*, it makes no sense to add a mass of methane emitted to a mass of carbon dioxide emitted, as these gases have different physical and chemical properties and thus different environmental impacts per unit of mass emitted). Therefore, a life cycle impact assessment (LCIA) method is needed to (1) aggregate the list of elementary flows into a much shorter and more manageable list of environmental "impact categories" and (2) "translate" the LCI results into meaningful information about the environmental impacts of the product system (Hauschild & Huijbregts, 2015).

Per ISO 14044, LCIA comprises three mandatory elements and four optional elements. The mandatory elements are (sec. 4.4.2.1):

> selection of impact categories, category indicators and characterization models;
>
> assignment of LCI results to the selected impact categories (classification); [and]
>
> calculation of category indicator results (characterization).

Regarding the selection of impact categories, best practice, following the requirements of ISO 14044, is to assess a comprehensive set of environmental impacts related to emissions to air, water, and soil. By integrating multiple types of environmental impacts (*e.g.*, climate change, acidification, air pollution, and ecotoxicological effects) through the full “life cycle” of a product system, LCA helps avoid “problem-shifting” that could result from a narrower focus on a limited subset of environmental issues and life cycle stages (Finkbeiner, 2009). For example, a widely cited (albeit outdated) comparative LCA of internal combustion engine vehicles (ICEVs) and electric vehicles (EVs) by Hawkins et al. (2012) indicates that although EVs may have lower global warming potential (GWP) over their life cycle (largely depending on the electricity supply mix for vehicle charging), they may also have higher impacts in terms of eutrophication, human and ecological toxicity, and mineral resource use. It is rarely, if ever, the case that one product is “better for the environment” in all respects.

As illustrated in Figure 2, the final two mandatory elements of LCIA – classification and characterization – concern the “environmental mechanism” from the LCI results (*e.g.*, greenhouse gas emissions, like carbon dioxide, methane, and nitrous oxide, and acidifying emissions, like sulfur dioxide and nitrogen oxides) to the “category endpoint” (*e.g.*, the ultimate effects of climate change and acidification on human health and the natural environment). Using a *characterization model*, the LCIA results for each impact category are expressed in terms of a *category indicator* which “…can be chosen anywhere along the environmental mechanism between the LCI results and the category endpoint(s)” (ISO 14044, sec. 4.4.2.2.2). LCA practitioners commonly use the term “midpoint” to refer to a category indicator positioned to partially model the environmental mechanism (*e.g.*, radiative forcing with respect to climate change, and proton release with respect to acidification), and the term “endpoint” to refer to a category indicator positioned at (in the terminology of ISO 14044) the category endpoint (*e.g.*, disability-adjusted life-years (DALYs) to represent harms to human health, and species loss to represent harms to natural ecosystems) (Hauschild & Huijbregts, 2015). There is a trade-off between the simpler interpretation of “endpoint” indicators and the simpler modelling (and thus relatively lower scientific uncertainty) of “midpoint” indicators (Hauschild & Huijbregts, 2015). In the characterization model, each LCI result (*i.e.*, each elementary flow) assigned to a given impact category is multiplied by a *characterization factor* that expresses the LCI results in common units of the category indicator (ISO 14044:2006, Hauschild & Huijbregts, 2015). For “midpoint” indicators, this is typically done using a reference substance (*e.g.*, the GWP, due to radiative forcing, of greenhouse gas emissions is commonly expressed as a mass of “carbon dioxide (chemical formula CO_2) equivalent” emissions, and the acidification potential, due to proton release, of acidifying emissions can be expressed as a mass of “sulfur dioxide (chemical formula SO_2) equivalent”) (ISO 14044:2006, Hauschild & Huijbregts, 2015). As can be seen in subsequent chapters of this thesis, the looser terminology of “environmental footprint” (*i.e.*, referring broadly to environmental emissions and their potential environmental impacts) and “carbon footprint” (*i.e.*, referring more narrowly to greenhouse gas emissions, typically expressed as a mass of “CO_2 equivalent” emissions – essentially corresponding to the GWP indicator used in LCA) is commonly used in practice (see, *e.g.*, Finkbeiner (2009), Ivanova et al. (2016), Jiménez-González et al. (2004), and Santero et al. (2011)), especially among those who are not LCA specialists (see, *e.g.*, Daughton & Ruhoy (2009), Lim et al. (2013), Panyakaew & Fotios (2011), Sáez-Almendros et al. (2013), and Talibi et al. (2022)).

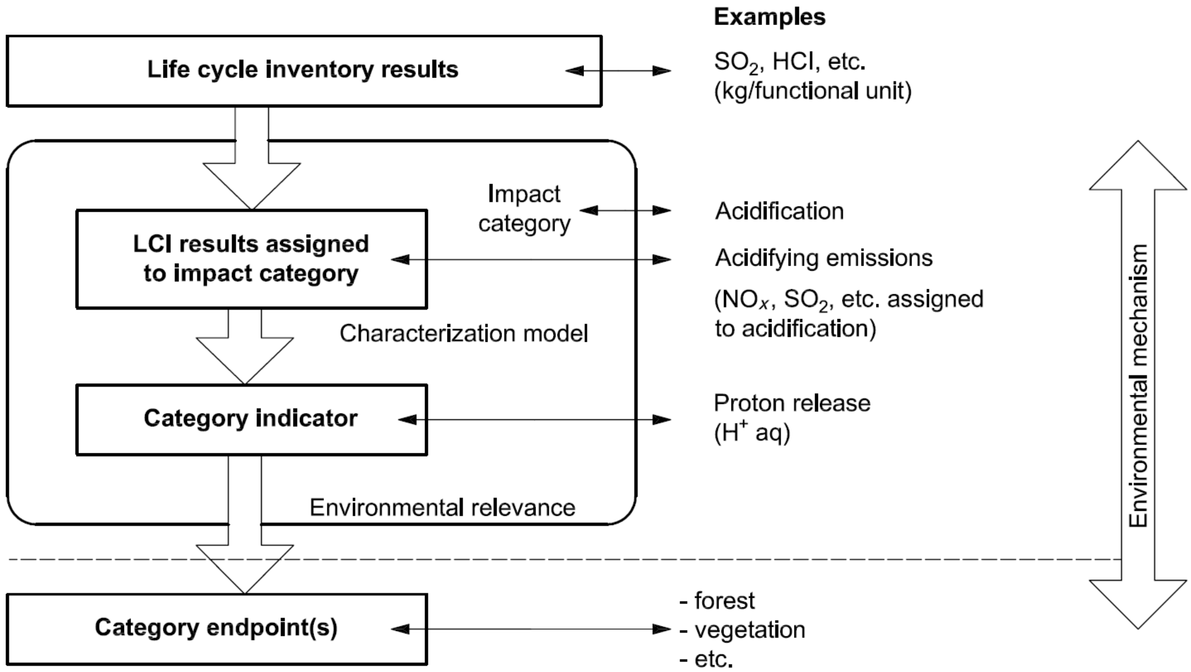


Figure 2: Life cycle impact assessment (ISO 14044:2006, Figure 3)

Per ISO 14044, the optional elements of LCIA are *normalization*, *grouping*, *weighting*, and *data quality analysis*. *Normalization* expresses a category indicator result in relation to a reference value; this is commonly done by defining a "baseline" scenario and expressing the category indicator results of an alternative scenario relative to the baseline (*e.g.*, expressing the global warming potential of an EV as a percentage of the global warming potential of an ICEV). *Grouping* essentially categorizes the impact categories – *e.g.*, into global, regional, or local impacts, or into high, medium, or low priority. *Weighting* is the aggregation of category indicator results based on explicit value-choices – as seen in LCIA methods, like the "Eco-indicator" (Goedkoop et al., 1998; Goedkoop & Spriensma, 2001) method and the "Ecological Scarcity" method (Frischknecht & Büsser Knöpfel, 2014), that express results in a single score of "points." Though understandably controversial (and prohibited by ISO 14044 for LCAs intended to be used in comparative assertions intended to be disclosed to the public), such value-choices – whether explicit or not – are ultimately necessary for decision-making; in my view, the key is to be *transparent* about the value-choices made and the rationale behind them. Per ISO 14044, data quality analysis can include *gravity analysis* (*i.e.*, through statistical procedures to highlight the most influential data points in the LCA), *uncertainty analysis* (*i.e.*, through statistical procedures to examine how the uncertainty in the LCA data and assumptions collectively affect the LCIA results) and *sensitivity analysis* (*i.e.*, to examine the effects of specific changes in data, methods, and assumptions – *e.g.*, by considering how different allocation[1] methods influence the LCI and LCIA results).

As illustrated in Figure 1, and probably evident from my explanation of LCA thus far, the interpretation phase of LCA takes place throughout the entire LCA process; although for simplicity and readability I have been walking through the LCA phases in a linear fashion, in practice LCA tends to be an iterative process. Per ISO 14044 (sec. 4.5.1.1), LCA interpretation comprises, among other things, the following three elements:

> identification of the significant issues based on the results of the LCI and LCIA phases of LCA;

[1] Allocation, defined by the international LCA standards (particularly ISO 14044:2006) as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (sec. 3.17), is a common – and hotly debated (Guinée et al., 2021) – methodological challenge in LCA. An "allocation problem" occurs when one process or product system produces multiple products, termed "co-products." The core problem is in finding a way to attribute, or *allocate*, a "fair" share of the resource use and emissions – and corresponding environmental impacts – of the multi-output process to the product of interest for the LCA. In practice, allocation is commonly done using the mass or economic values of the co-products, though these approaches have been criticized for introducing unwanted variability (as economic values fluctuate over time) and failing to provide good proxies for the phenomenon of interest (*i.e.*, as mass or economic values are not necessarily good predictors of environmental impacts) – see, *e.g.*, Pelletier et al. (2015), Pelletier & Tyedmers (2011, 2012), and Weinzettel (2012).

an evaluation that considers completeness, sensitivity and consistency checks; [and]

conclusions, limitations, and recommendations.

"Identification of significant issues" typically takes the form of *contribution analysis* (Heijungs et al., 2005) to highlight "hotspots" in the studied product system(s); "hotspots" are those parts of the system (*e.g.*, life cycle stages, product components, and unit processes) that make the largest contributions to the LCI or LCIA results (Hellweg & Mila i Canals, 2014). Experience has shown that certain types of products tend to have characteristic "life cycle profiles" based on the biggest "hotspots" in their life cycle (*e.g.*, automobiles, buildings, and appliances tend to have the largest contributions from the "use" of these products, whereas furniture tends to have the largest contributions from the production of the materials used to make the furniture); see, *e.g.*, the works of Young (1996) and Ashby (2013) for further elaboration and examples of the concept of "life cycle profiles."

Per ISO 14044, the evaluation element includes, at a minimum, a completeness check, a sensitivity check, and a consistency check (sec. 4.5.3.1). The completeness check verifies that all relevant data and information needed for the LCA, in accordance with the goal and scope definition, are available and complete. The sensitivity check evaluates the reliability of the results and conclusions of the LCA, considering uncertainties tied to the data, methods, and assumptions used. Results of uncertainty analysis and sensitivity analysis, if conducted in the LCI or LCIA phase, are to be included in the sensitivity check (ISO 14044, sec. 4.5.3.3). Sensitivity analysis is especially important – and explicitly required by ISO 14044 – for LCAs intended to be used in comparative assertions intended to be disclosed to the public. The consistency check verifies that the data, methods, and assumptions used are consistent throughout the LCA and are consistent with the goal and scope definition.

Finally, conclusions are drawn (with consideration of study limitations), and recommendations are made to the intended audience(s) in accordance with the goal and scope definition.

1.4 Thesis overview

This chapter has established the conceptual foundations for my thesis; in this context, the remainder of the thesis is structured as follows. In Chapter 2 (co-authored with Jair Santillán-Saldivar, Cassandra Thiel, Guido Sonnemann, and Steven B. Young), I conduct a comprehensive scoping review of the literature on “healthcare sustainability” – based on a representative sample from over 1,700 articles published between 1987 and 2017 – that highlights largely untapped opportunities to use industrial ecology approaches (such as LCA) to build an evidence base for this burgeoning domain of sustainability management. In Chapter 3 (co-authored with Steven B. Young), I constructively critique the existing methodology for *organizational* LCA (O-LCA), with concrete proposals – particularly the use of basic statistical sampling and inference techniques – to strike a better balance of scientific rigour and practical feasibility in O-LCA. In Chapter 4 (co-authored with Steven B. Young), I test and demonstrate these proposals through an O-LCA of a Canadian hospital, in which I compiled new LCA data for approximately 200 goods and services used in healthcare. Finally, in Chapter 5, I conclude by reflecting upon my contributions in this thesis, and by making suggestions for future work.

Chapter 2

2. Potential for industrial ecology to support healthcare sustainability: Scoping review of a fragmented literature and conceptual framework for future research

The contents of this chapter are published in:

Cimprich, A., Santillán-Saldivar, J., Thiel, C.L., Sonnemann, G., & Young, S.B. (2019). Potential for industrial ecology to support healthcare sustainability: Scoping review of a fragmented literature and conceptual framework for future research. *Journal of Industrial Ecology, 23(6)*, 1344-1352. DOI: 10.1111/jiec.12921

2.1 Summary

Healthcare is a critical service sector with a sizable "environmental footprint" from both direct activities and the indirect emissions of related products and infrastructure. As in all other sectors, the "inside-out" environmental impacts of healthcare (*e.g.*, from greenhouse gas emissions, smog-forming emissions, and acidifying emissions) are harmful to public health. The environmental footprint of healthcare is subject to upward pressure from several factors, including the expansion of healthcare services in developing economies, global population growth, and aging demographics. These factors are compounded by the deployment of increasingly sophisticated medical procedures, equipment, and technologies that are energy- and resource-intensive. From an "outside-in" perspective, on the other hand, healthcare systems are increasingly susceptible to the effects of climate change, limited resource access, and other external influences. We conducted a comprehensive scoping review of the existing literature on environmental issues and other sustainability aspects in healthcare, based on a representative sample from over 1,700 articles published between 1987 and 2017. To guide our review of this

fragmented literature, and to build a conceptual foundation for future research, we developed an industrial ecology framework for healthcare sustainability. Our framework conceptualizes the healthcare sector as comprising "foreground systems" of healthcare service delivery that are dependent on "background product systems." By mapping the existing literature onto our framework, we highlight largely untapped opportunities for the industrial ecology community to use "top-down" and "bottom-up" approaches to build an evidence base for healthcare sustainability.

2.2 Introduction

Healthcare is a critical service sector, typically representing a large portion of spending in developed economies, with a sizable "environmental footprint"[2] from both direct activities and the indirect emissions of related products and infrastructure. The combined direct and indirect greenhouse gas emissions from healthcare have been estimated at 10% of the national total in the United States (Eckelman & Sherman, 2016), 7% in Australia (Malik et al., 2018), 5% in the United Kingdom (NHS Sustainable Development Unit, 2016), and 4.6% in Canada (Eckelman et al., 2018).

[2] The term "environmental footprint" is a common shorthand (see, *e.g.*, Daughton & Ruhoy (2009), Ivanova et al., (2016), Jiménez-González et al. (2004), Lim et al. (2013), Panyakaew & Fotios (2011), Sáez-Almendros et al. (2013), Santero et al. (2011), and Talibi et al. (2022)) for the more technically precise terminology, codified in international standards (ISO 14040:2006, ISO 14044:2006) used in environmental life cycle assessment (LCA), particularly the terms "emissions" (*e.g.*, greenhouse gas emissions, other air emissions, emissions to water, and emissions to soil) and "potential environmental impacts" (*e.g.*, global warming potential, acid precipitation potential, smog formation potential, and eutrophication potential). A related term, "carbon footprint," is widely used to refer to greenhouse gas emissions (Finkbeiner, 2009), typically expressed in units of global warming potential (*i.e.*, as a mass of "carbon dioxide (chemical formula CO_2) equivalent" emissions).

Using terminology from influential business management scholars (Porter & Kramer, 2006), we conceptualize the environmental impacts of healthcare (*e.g.*, from greenhouse gas emissions, smog-forming emissions, and acidifying emissions) as “inside-out” impacts. As in all other sectors, these “inside-out” impacts are harmful to public health. According to the aforementioned study by Eckelman & Sherman (2016), the environmental emissions from the US healthcare sector are responsible for the loss of up to 470,000 disability-adjusted life years (DALYs) annually (405,000 DALYs when adjusted for reductions in the carbon intensity of electricity generation). These figures are on par with the number of deaths from preventable medical errors in the US healthcare sector (Eckelman & Sherman, 2016). The environmental footprint of healthcare is subject to upward pressure from several factors. These include the expansion of healthcare services in developing economies, global population growth (the United Nations Department of Economic and Social Affairs (2017) projects a global population of almost 10 billion by 2050), and aging demographics (according to the same UN report). These factors are compounded by the deployment of increasingly sophisticated medical procedures, equipment, and technologies that are energy- and resource-intensive.

On the other hand, healthcare systems are increasingly susceptible to the effects of climate change (*e.g.*, property damage from extreme weather events and growing demand for treatment of heat-related illnesses), limited resource access (*e.g.*, due to supply disruptions of “critical raw materials” like specialized metals and alloys used in complex medical devices and equipment), and other external influences. Based on the terminology from Porter & Kramer (2006), we conceptualize these external influences as “outside-in” impacts *on* healthcare.

There is a growing research and policy interest regarding environmental issues and other sustainability aspects in healthcare (*e.g.*, in the World Health Organization (2017), the World

Bank (2017), and the Lancet Commission on Climate Change and Health (Watts et al., 2017)). In April 2018, the Workshop on Environmental Sustainability in Clinical Care, co-hosted by Yale University and New York University (henceforth referred to as "the Yale Workshop"), brought together experts from around the world, including three of the authors[3], to discuss the state of the art and highlight future research directions (Thiel et al., 2018). At the Yale Workshop, we presented preliminary results from the work contributed in this article—a comprehensive scoping review of the existing literature on healthcare sustainability, based on a representative sample from over 1,700 articles published between 1987 and 2017. Although most of the literature we collected is focused on environmental and resource-related issues, we aimed at capturing a wide range of sustainability aspects, including economic and social dimensions to a limited extent.

This article is structured as follows. We begin by explaining the methodology of our scoping review—a relatively new type of literature review that is similar to, but not the same as, a systematic review—including the literature search protocol, inclusion criteria, and content analysis. In a departure from the conventional scoping review methodology, we incorporate a novel application of a basic statistical inference technique—stratified random sampling—to obtain a representative sample of 157 articles for review. To guide content analysis of the existing literature, and to build a conceptual foundation for future research, we developed an industrial ecology framework for healthcare sustainability. Our framework conceptualizes the healthcare sector as comprising "foreground systems" of healthcare service delivery that are dependent on "background product systems." By mapping the existing literature onto our framework, we highlight largely untapped opportunities for the industrial ecology community to

[3] Alexander Cimprich, Cassandra L. Thiel, and Steven B. Young

use “top-down” and “bottom-up” approaches to build an evidence base for healthcare sustainability.

2.3 Methods

To build a comprehensive overview of the existing literature on healthcare sustainability, we applied a variation of a relatively new literature review methodology—the *scoping review*. Scoping reviews are similar to, but not the same as, systematic reviews. Although both methods follow a systematic protocol for collecting and analyzing relevant literature, they serve different purposes. Systematic reviews aim to synthesize the collective body of evidence regarding a specific question(s)—for example, the efficacy of a particular medical intervention (Peters et al., 2015) or the life cycle environmental impacts of a specific product type (Zumsteg et al., 2012). Scoping reviews are more exploratory in nature, with the aim of “mapping” broad fields of literature, highlighting gaps in the existing knowledge base, and prioritizing directions for future research (Peters et al., 2015). This was precisely our aim with respect to the burgeoning field of healthcare sustainability.

Following the scoping review methodology developed by Peters et al. (2015) and outlined in Figure 3, we conducted systematic literature searches via two large international databases: Scopus and PubMed. Although most of the literature we collected is focused on environmental and resource-related issues, we aimed at capturing a wide range of sustainability aspects, including economic and social dimensions to a limited extent (see Supporting Information S1 on the Journal’s website). We excluded broader considerations of wellbeing, social equity, access to healthcare, and financial sustainability. These topics are important, but beyond the scope of our industrial ecology framework. Search phrases were constructed by combining “sustainability-related” keywords with “healthcare-related” keywords. “Sustainability-related” keywords included generic terms like *environment*, *sustainability*, and *social responsibility,* along with more specific keywords like *carbon footprint*, *energy efficiency*, *environmental management*, *waste management*, *life cycle assessment*, and *material flow analysis*. Boolean operators were used to capture synonyms (*e.g.*, *carbon footprint OR greenhouse gas* OR climate change OR global warming*) and related words (*e.g.*, *sustainab** captures terms like *sustainable*, *sustainability*, and *sustainable development*). Similarly, “healthcare-related” keywords included generic terms (*e.g.*, *healthcare OR health care*, *medic**, *clinic**, *surger* OR surgical*) and specific keywords (*e.g.*, *X-ray*, *ultrasound*, *magnetic resonance imaging*, *medical device**, *long-term care*, *prevent* care,* and *pharmaceutical**). When searching the PubMed database, which is oriented toward the health sciences and medicine, only “sustainability-related” keywords were used. Using the PubMed database helped capture any relevant literature we may have missed when searching the Scopus database. In all, we conducted 134 literature searches between September 29 and November 17, 2017 (details in Supporting Information S2).

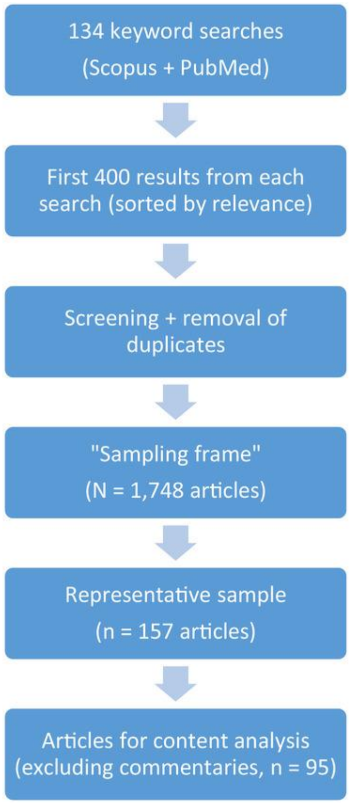


Figure 3: Scoping review flow diagram

We screened the first 400 results from each search (sorted by relevance) according to the following inclusion criteria. First, our review was limited to academic journal articles published from 1987 onward (*i.e.*, corresponding to the landmark "Brundtland report" and available in English (World Commission on Environment and Development [WCED], 1987). Although the lack of "grey literature" in our review is a limitation, restricting our scope to academic journal articles provided us with a "sampling frame" comprising documents of similar format (*i.e.*, every article we reviewed has an academic journal title attached to it). This enabled us to analyze basic characteristics of the *academic* literature on healthcare sustainability (*e.g.*, the distribution of publications between journal titles and between the subject matter classifications in the Scopus database). It also facilitated a novel application of a basic statistical inference technique—

stratified random sampling—to our scoping review methodology (as further explained in the following paragraphs). To keep our scope limited to healthcare services provided for *human* patients, we excluded literature on veterinary care. We included editorials and commentary articles, as these are part of the healthcare sustainability discourse—especially in medical and health science journals. We did not, however, conduct full content analysis on these documents, as they do not make substantive contributions to healthcare sustainability research.

Screening of search results based on our inclusion criteria, along with removal of duplicates, yielded a list of 1,748 articles. Although conventional systematic and scoping review protocols include separate steps for screening and removal of duplicates, given the broad coverage of literature we aimed for, and the large number of literature searches we conducted (134 in total), we conducted these steps in parallel. We imported search results from the Scopus and PubMed databases into a spreadsheet, in which we removed duplicates along with search results that did not meet our inclusion criteria (*e.g.*, book chapters, non-academic articles, articles not available in English, and articles published before 1987). We do not see the lack of separate steps as a limitation, as duplicates must be removed in any case, and they have no bearing on our inclusion criteria or on the final results of our scoping review. Moreover, our literature search protocol (see details of keyword searches in Supporting Information S2) and inclusion criteria (as outlined above, with further details on "sustainability aspects" in Supporting Information S1) are described in sufficient detail to be replicable by an independent researcher.

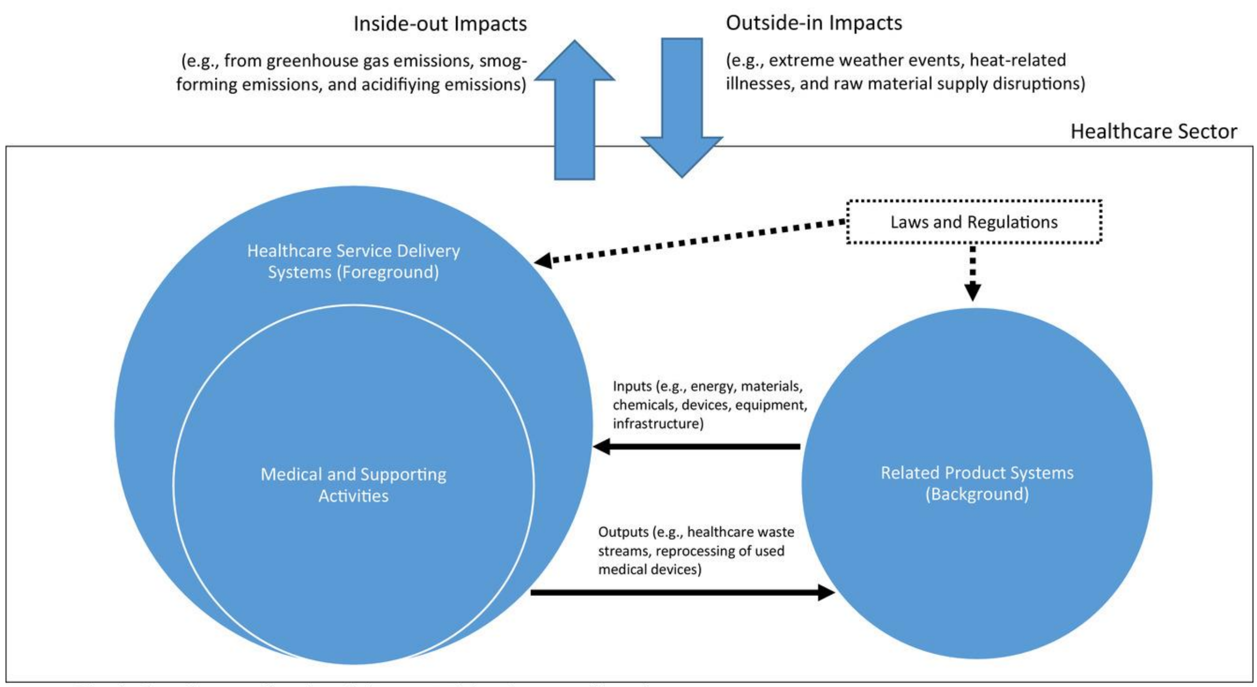
As mentioned earlier, our biggest departure from the conventional scoping review methodology was our novel application of a basic statistical inference technique—stratified random sampling—to representatively understand the content of the literature we collected based on a relatively small subset of this literature. It is impractical to read over 1,700 articles—especially given the time-sensitive nature of a systematic or scoping review (*i.e.*, new literature continues to be published while the review is conducted, thus diminishing the timeliness and value of the review). The conventional approach is to limit the scope of the review through the inclusion and exclusion criteria (Peters et al., 2015). For example, Godfrey et al. (2013) conducted a scoping review of "homecare safety and medication management with older adults." The scope of the review was limited by participant characteristics (*i.e.*, "older adults," defined as individuals aged 65 and older), key concepts (*i.e.*, medication management), and contextual factors (*i.e.*, in-home care). Following the initial screening of search results based on the *inclusion* criteria, the conventional scoping review methodology described by Peters et al. (2015) also includes a step in which additional *exclusion* criteria are applied to narrow the collected literature down to a manageable selection of relevant articles for review.

Our review was significantly broader than a conventional systematic review, or even a conventional scoping review. The 1,748 articles we collected through our search protocol comprised our literature "sampling frame." Theoretically, the "population" for our review would have comprised all relevant literature on healthcare sustainability. Our sampling frame is an imperfect, but reasonably robust, representation of this literature population, subject to the limitations of our search protocol—including the coverage of the databases searched, the keyword search phrases used, and the inclusion criteria applied.

Within our literature sampling frame, we applied a stratified random sampling approach using journal "subject clusters" in the Scopus database: life sciences, social sciences, physical sciences, and health sciences. To ensure that each article was classified into only one subject cluster, we created a fifth subject cluster, termed "cross-disciplinary," to capture journals classified under more than one of the first four subject clusters. We assigned each of the 1,748 articles a random number, and then sorted the articles by these numbers (in ascending order). We obtained a representative sample of 157 articles by selecting articles (in the randomized order) from each subject cluster (serving as a "stratum" in sampling terminology) such that the distribution of articles between subject clusters reflected that of our "sampling frame" of 1,748 articles.

Although it cannot overcome the limitations of our literature sampling frame as discussed in the preceding paragraphs, random sampling is a well-established, scientifically sound technique (as described in statistics textbooks such as DeVeaux et al. (2015)) that supports statistical *inference* about the sampling frame without the need to review all the literature contained within it. Randomization avoids sampling bias and ensures that, to a specified degree of uncertainty, the sample is representative of the sampling frame from which it is drawn. *Stratifying* the sample according to identifiable "sub-populations" (*e.g.*, journal "subject clusters" in our case) improves upon simple random sampling by ensuring that these "sub-populations" are proportionally represented in the sample (DeVeaux et al., 2015). Supporting Information S3 describes our literature sampling frame in terms of the distribution of publications over time and between the subject matter classifications in the Scopus database. It can be seen that the existing healthcare sustainability literature is highly fragmented—being spread out over hundreds of academic journals in a wide variety of fields. Over 700 articles were published in journals with fewer than three relevant articles each. As discussed in the Yale Workshop, this fragmentation makes it difficult to target appropriate audiences for the dissemination of healthcare sustainability research. It can also complicate the publishing process, as reviewers and journal editors—particularly in medicine—may not be familiar with this burgeoning field.

To guide content analysis on our literature sample of 157 articles, and to build a conceptual foundation for future research, we developed an industrial ecology framework for healthcare sustainability. We refined our framework based on feedback from participants at the Yale Workshop, as well as discussions with colleagues in the fields of medicine and healthcare management. As illustrated in Figure 4, our framework conceptualizes the healthcare sector as comprising "foreground systems" of healthcare service delivery (*e.g.*, hospital campuses, ambulatory surgery centers, clinics, and laboratories) in which medical activities (*e.g.*, diagnostic imaging and operating room procedures) and "supporting" activities (*e.g.*, building maintenance, hospital food services, and healthcare waste management) take place (details in Supporting Information S1).



Physical exchanges (i.e., involving material and energy flows)

Non-physical influences (e.g., laws and regulations pertaining to healthcare waste management, environmental protection, workplace health and safety, and infection control)

Figure 4: Industrial ecology framework for healthcare sustainability

The “foreground systems” have physical exchanges with “background product systems.” On the input side, background product systems supply energy (in the form of electricity and fuels), materials, chemicals (including active pharmaceutical ingredients and compounds), devices, equipment, and infrastructure used to provide healthcare services. On the output side, background product systems facilitate “end-of-life” management (*e.g.*, through contracted waste management services and reprocessing of used medical devices).

Our framework incorporates the “inside-out” impacts of foreground and background systems (*e.g.*, from greenhouse gas emissions, smog-forming emissions, and acidifying emissions), along with the “outside-in” impacts of external influences on these systems (*e.g.*, property damage from extreme weather events, growing demand for treatment of heat-related illnesses, and supply disruptions of critical raw materials like specialized metals and alloys used in complex medical devices and equipment).

Finally, our framework recognizes that foreground and background systems are subject to the legal and regulatory contexts of the jurisdictions in which they operate. Healthcare-related laws and regulations (*e.g.*, pertaining to critical issues like healthcare waste management, environmental protection, workplace health and safety, and infection control) vary from country to country (as well as within countries), and are a key influence on physical flows associated with healthcare (as seen, *e.g.*, in the literature on healthcare waste management and the trend toward single-use medical devices and supplies).

As illustrated in Figure 5, and further discussed in the next section, we mapped the existing healthcare sustainability literature onto our framework by coding each of the 95 research articles (*i.e.*, excluding the 62 articles we coded as “commentaries”) in our literature sample according to categories of medical activities, supporting activities, and background product systems. We also coded the articles based on the “inside-out” and/or “outside-in” sustainability aspects addressed. Coding the articles in this way enabled us to analyze the distribution of literature in relation to our framework (*i.e.*, the percentage of articles coded in each category). As explained earlier, our literature sample was designed to be representative of our full “sampling frame” of 1,748 articles collected through our literature search protocol. We therefore *inferred* the “true” distribution of literature based on our sample. As described in Supporting Information S4, we constructed 95% confidence intervals for the percentages. Precise descriptions of our coding categories are provided in Supporting Information S1. A complete list of the 157 articles analyzed, with bibliographic information and coding, is provided in Supporting Information S5.

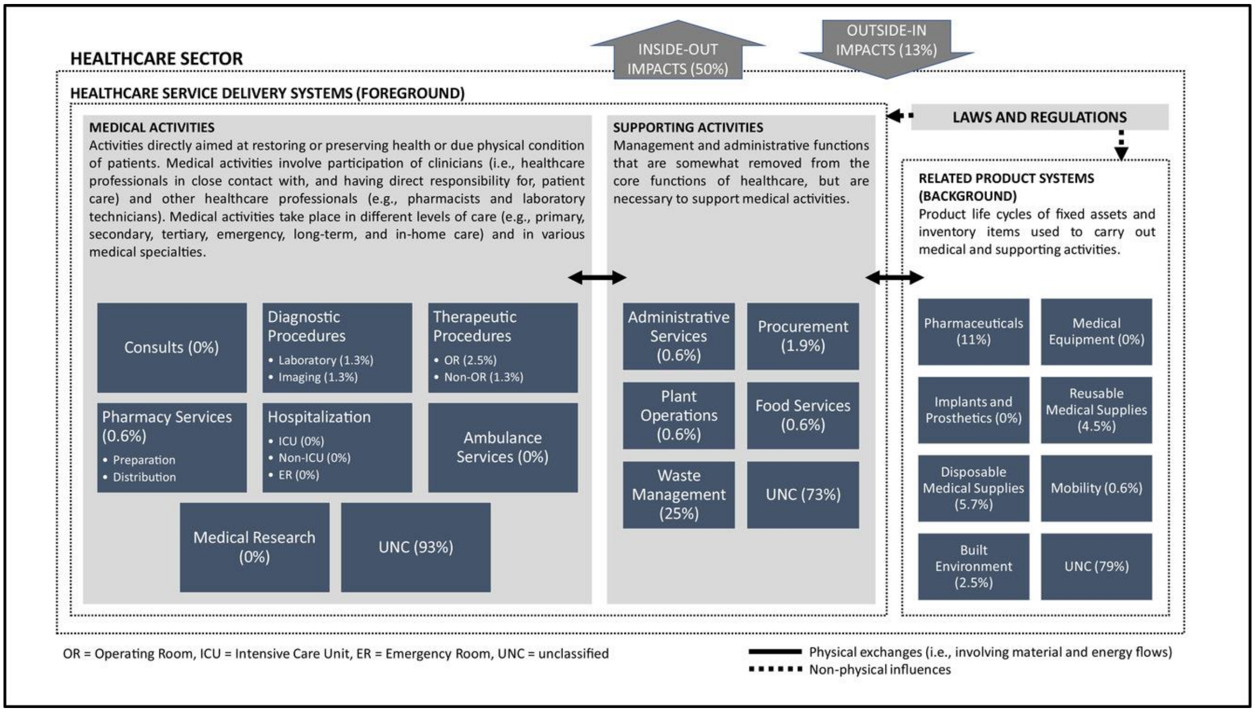


Figure 5: Distribution of healthcare sustainability literature (n = 157 articles) in relation to our industrial ecology framework.[4]

2.4 Results

Through the lens of our industrial ecology framework, two major gaps are evident in the existing healthcare sustainability literature. First, many components of our framework, including important medical activities, supporting activities, and background product systems, are

[4] The "true" distribution of literature (*i.e.*, the actual percentages [of N = 1,748 articles] in each category) was *inferred* from the sample. For simplicity, margins of error (provided in Supporting Information S4) are omitted from this figure. For several reasons, the percentages do not sum to 100%. First, coding categories (including categories of medical activities, supporting activities, background product systems, and "inside-out" vs. "outside-in" impacts) are not mutually exclusive (*i.e.*, the same article may be coded under multiple categories). Second, the percentages of "inside-out" and "outside-in" impacts shown here are totals out of our sample of 157 articles; they are not tied to any categories of medical activities, supporting activities, or background product systems. Finally, 62 articles were coded as "commentaries" and excluded from content analysis; therefore, these articles are not captured in the "inside-out" and "outside-in" percentages. A value of 0% does not necessarily mean that there is no literature in that category (although that is a possibility); it means that there is no such literature *in our sample*.

understudied from a sustainability perspective. Second, the existing literature is limited by the narrow scope of sustainability aspects addressed.

As can be seen in Figure 5, there is relatively little literature that explicitly examines sustainability aspects of specific medical activities. Exceptions include studies of energy use in diagnostic imaging (N. P. Burke & Stowe, 2015; Esmaeili et al., 2015), which constitute about 1.3% (±1.8 percentage points[5]) of the 1,748 articles we collected, along with a growing interest in LCA studies of operating room procedures (*e.g.*, a study of hysterectomies by Thiel et al. (2015) and a study of birthing procedures by Campion et al. (2012)), which constitute about 2.5% (±2.5 percentage points). There is a paucity of literature regarding sustainability aspects of consults, pharmacy services, hospitalization, ambulance services, and medical research. These activities may not be equally important from a sustainability perspective (*e.g.*, consults are likely to be less resource- and energy-intensive than operating room procedures), but without further study, this is difficult to gauge. Indeed, part of the purpose of industrial ecology tools, like LCA and economic input–output analysis, is to highlight “hotspots” of resource use and environmental impacts. Based on the existing literature, the “hotspots” of healthcare are not well understood. Of particular concern is the lack of literature on pharmacy services, inpatient hospitalization (particularly in emergency rooms and intensive care units), and ambulance services. These are important medical activities, and their sustainability aspects are probably not trivial.

With the notable exception of healthcare waste management (25% ± 6.8 percentage points), “supporting” activities are also understudied. The literature on healthcare waste management

[5] In this case, the margin of error is larger than the estimated percentage of articles in this category. With 95% confidence, the “true” percentage (*i.e.*, of all 1,748 articles in our literature sampling frame) could be as large as 3.1% (*i.e.*, 1.3% + 1.8%), or it could be close to 0%).

includes, but is not limited to, reviews of legal and regulatory requirements (Botelho, 2013; Haylamicheal & Desalegne, 2012; Takatsuki, 2000), surveys and case studies of healthcare waste management attitudes and practices (Aseweh Abor & Bouwer, 2008; Askarian et al., 2004; S. C. Gupta et al., 2014; Idowu et al., 2013; Jovanović et al., 2016; Saad, 2013; Thiel, Duncan, et al., 2017; Ul Rahman et al., 2017), waste audits (Majid & Umrani, 2006; Patil & Pokhrel, 2005; Patwary et al., 2009; Saad, 2013; Suwannee, 2002), and toxicological studies of healthcare wastes (P. Gupta et al., 2009). The evident focus on waste management probably reflects the visibility of this issue, the large volumes of waste generated in healthcare services (*e.g.*, the US healthcare sector generates about 1.7 million tons of solid waste annually [United States Environmental Protection Agency, 2005]), and the complex legal and regulatory aspects related to medical waste.

With regard to "background product systems," the categories with the most coverage in the literature include pharmaceuticals (11% ± 4.9 percentage points), reusable medical supplies (4.5% ± 3.2 percentage points), and disposable medical supplies (5.7% ± 3.6 percentage points). Sustainability aspects of active pharmaceutical ingredients and compounds are related to production of these chemical products (Raymond et al., 2010; Van der Vorst et al., 2010, 2011) and their effects when released into the environment (Derksen et al., 2004; Straub, 2016). The literature on reusable and disposable medical supplies includes comparative LCAs, for example, of disposable and reusable laryngeal mask airways (Eckelman et al., 2012) and of metered dose inhalers and electric nebulizers (Goulet et al., 2017). There is a paucity of literature on production of medical equipment, materials and manufacturing of implants and prosthetics, and mobility of patients and staff.

The existing literature on healthcare sustainability is also limited by its narrow coverage of sustainability aspects. As can be seen in Figure 5, the "inside-out" impacts *of* healthcare (50% ± 7.8 percentage points) have received significantly more coverage than "outside-in" impacts *on* healthcare (13% ± 5.2 percentage points). Supporting Information S4 provides a more detailed analysis of the sustainability aspects addressed in relation to medical activities, supporting activities, and background product systems. Most of these sustainability aspects (*e.g.*, common environmental LCA impact categories like acidification, eutrophication, and ecotoxicity) can be considered "inside-out," though some aspects, like climate change, can be either "inside-out" (*i.e.*, greenhouse gas emissions from healthcare) or "outside-in" (*i.e.*, effects of climate change like extreme weather events and heat-related illnesses). Overall, energy use (11% ± 4.9 percentage points) and climate change (18% ± 6.0 percentage points) have received the most coverage of all sustainability aspects considered in our review. While energy use and climate change are important issues, other environmental and social aspects, like smog-forming emissions, acidifying emissions, and labor practices in healthcare supply-chains, are often overlooked. "Outside-in" impacts, particularly in terms of resource-related issues like supply disruption risks of critical raw materials, are also underexplored.

2.5 Discussion and conclusions

We have conducted a timely and comprehensive scoping review of the existing literature on healthcare sustainability. Previous limited reviews have been conducted in this space—for example, a review of recycling and waste management practices by Kwakye et al. (2011), a review of environmental sustainability in hospitals (mostly focused on what are considered "foreground" activities in our framework) by McGain & Naylor (2014), a review of pharmaceutical sustainability by De Soete et al. (2017), and a review of environmental considerations in health technology assessment by Polisena et al. (2018). Nonetheless, to the best of our knowledge, our review is the broadest survey to date—using a wide range of keyword searches—of this fragmented literature.

There are limitations to our review. First, our "sampling frame" of 1,748 articles is an imperfect representation of the existing literature on healthcare sustainability, subject to the limitations of the databases searched (no database is fully comprehensive) and the keyword search phrases used (which reflect our limited perspective, knowledge, and understanding). Given that we limited our scope to academic journal articles, "grey literature," such as the carbon footprinting reports of the UK NHS Sustainable Development Unit, was not covered. Our review was also limited to a static snapshot of literature published between 1987 and 2017, with the last search conducted on November 17, 2017. Consequently, our review does not cover the most recent advancements in this burgeoning field.

Second, our review does not provide complete coverage of our literature sampling frame. Content analysis was conducted on only a relatively small subset of the 1,748 articles we collected through our literature search protocol. When interpreting the results of our literature "mapping" analysis (*i.e.*, the category percentages in Figure 5), a value of 0% does not necessarily mean that there is no literature in that category (although that is a possibility); it means that there is no such literature *in our sample*. Nonetheless, our stratified random sampling design avoids sampling bias and ensures that the 157 articles we reviewed are *representative* of the 1,748 articles in our sampling frame. Therefore, the "true" distribution of literature (*i.e.*, the actual percentages of articles in each category) can be *inferred* from the sample (subject to the margins of error).

Limitations notwithstanding, our review highlights largely untapped opportunities for the industrial ecology community to use “top-down” and “bottom-up” approaches to build an evidence base for healthcare sustainability. “Top-down” approaches can begin with a high-level overview of national or regional healthcare sectors to prioritize “hotspots” for deeper investigation. In their economic input–output analysis of the US healthcare sector, Eckelman & Sherman (2016) assessed the contributions of greenhouse gas emissions by National Health Expenditure (NHE) categories and economic sectors. The largest contributors by NHE category were Hospital Care (36%), Physician and Clinical Services (12%), and Prescription Drugs (10%). Contribution analysis by sector revealed that only 2.5% of greenhouse gas emissions were directly attributable to the activities of healthcare facilities (*e.g.*, from onsite boilers). The overwhelming majority of healthcare’s carbon footprint was attributed to the “embodied” emissions of purchased energy, goods, and services—termed “background product systems” in our framework. Similar observations were made for other environmental impact categories, and for the Canadian healthcare sector (Eckelman et al., 2018). Starting from this highly aggregated analysis, other industrial ecology tools, like material flow analysis and LCA, could be used to “zoom in” on healthcare service delivery systems (*e.g.*, hospital campuses), and the activities conducted therein (many of which are evidently understudied based on our review). There is also considerable scope for further investigation of background product systems (*e.g.*, LCAs of medical equipment, implants, and prosthetics).

Using “bottom-up” approaches, relatively granular LCAs of medical activities, supporting activities, and background product systems can be scaled up to “feed” into large-scale sector overviews like those conducted by Eckelman & Sherman (2016) and Eckelman et al. (2018). To our knowledge, large-scale healthcare footprinting studies have only been conducted for a limited number of developed countries—the United States (Eckelman & Sherman, 2016), the United Kingdom (NHS Sustainable Development Unit, 2016), Australia (Malik et al., 2018), and Canada (Eckelman et al., 2018)—and are often limited to carbon footprints rather than a wide range of environmental impact categories. It would be valuable to conduct similar studies in other parts of the world (*e.g.*, in the EU, Japan, and developing countries), to broaden the scope of sustainability aspects addressed (including "inside-out" and "outside-in" impacts), and to update the analyses as new data become available.

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Supplementary content

See Appendix A.

Chapter 3

3. Improving organizational LCA

The contents of this chapter are submitted for publication in:

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After the submission to the journal, and the defence of this thesis, some revisions have been made in this Chapter to further improve clarity, transparency, and overall scholarly quality. The revisions do not impact the core methodology, findings, or contribution of the manuscript.

3.1 Summary

Organizational LCA (O-LCA) is a new form of LCA that combines information from product LCAs to assess the environmental impacts of a larger, multi-product system. By scaling the LCA framework to aggregate many products in one study, while retaining LCA's core strengths – particularly its granularity and capacity to integrate multiple environmental impacts – O-LCA has potential to provide an even more powerful decision-making tool for sustainable development. However, there is limited academic literature on O-LCA, and the current state of the art – the O-LCA Guidance developed through the Life Cycle Initiative hosted by the United Nations Environment Programme (which builds upon the relevant ISO standards) – has methodological shortcomings not addressed in the existing literature. We therefore provide a constructive critique of the O-LCA Guidance, focusing on the LCA phases of goal and scope definition, life cycle inventory, and interpretation. Our biggest criticism concerns the flawed treatment of uncertainty in O-LCA; previous O-LCAs – following the Guidance – have used small, non-randomized sets of "representative" products to model broad and diverse product portfolios. We propose instead to use basic statistical sampling and inference techniques to strike

a better balance of scientific rigor and practical feasibility in O-LCA. We elaborate on our novel methodology in this article, and we refer to a forthcoming case study in which we test and demonstrate the approach.

3.2 Introduction

LCA is a widely used and internationally standardized approach for systematically evaluating the environmental impacts of a product system (ISO 14040:2006, ISO 14044:2006). This approach, facilitated through commercially available and/or open-source software and extensive databases, has several strengths that make it a powerful tool to support decision-making. First, it can integrate multiple types of environmental impacts (*e.g.*, climate change, acidification, air pollution, and ecotoxicological effects) through the full "life cycle" of a good or service system (*i.e.*, from raw material acquisition, through product manufacturing, product use, and product end-of-life). This whole-system perspective is important for avoiding "problem-shifting" that could result from a narrower focus on a limited subset of environmental issues or life-cycle stages – as highlighted in an editorial on "carbon footprinting"[6] by Finkbeiner (2009). Second, LCAs tend to be highly granular, using detailed data for unit processes (*e.g.*, resource extraction, materials production, and manufacturing processes) in the studied product system. This granularity provides the transparency needed to interpret and evaluate the methodology and results of an LCA, while facilitating the identification of environmental "hotspots" to prioritize areas for improving the environmental performance of the product system. Finally, the concept

[6] The term "carbon footprint" is widely used to refer to greenhouse gas emissions, typically expressed in units of global warming potential (*i.e.*, as a mass of "carbon dioxide (chemical formula CO_2) equivalent" emissions).

of the *functional unit* enables comparisons of environmental impacts between alternative product systems.

Given these strengths, and given the development of data and the power of software, one thrust of ongoing methodological and practical development in the LCA community is aimed at scaling the life cycle approach to examine larger, multi-product systems. Examples include organizational LCA (*i.e.*, LCA of an organization's operations over a given time period (Martínez-Blanco et al., 2015b; UNEP, 2015)), diet LCA (*i.e.*, LCA of all food products consumed in a person's diet (Baroni et al., 2007; Carlsson-Kanyama et al., 2003; Hallström et al., 2015; Heller et al., 2013; Hoolohan et al., 2013; Meier & Christen, 2013; Muñoz et al., 2010; Pairotti et al., 2015; Saxe et al., 2013; van Dooren et al., 2014; Veeramani et al., 2017), and "life-LCA" (*i.e.*, LCA of the life of a human being (Bossek et al., 2021; Goermer et al., 2020)). Organizational LCA (O-LCA) is, relatively speaking, the most established of these approaches, having benefited from standardization through an extension of the international standards on product LCA (ISO 14072:2014), the development of further methodological *Guidance* through the Life Cycle Initiative (hosted by the United Nations Environment Programme; UNEP (2015)), and the lessons learned through the experiences of 12 organizations in "road testing" the O-LCA Guidance (also through the Life Cycle Initiative (UN Environment, 2017)).

As illustrated in Figure 6, O-LCA can conceptually be situated in relation to other widely used analytical approaches for environmental sustainability – particularly product LCA, organizational carbon footprinting (*e.g.*, using the *Greenhouse Gas (GHG) Protocol Corporate Standard* (WRI & WBCSD, 2004)), and environmentally extended input-output (EEIO) LCA – in terms of the granularity of data and results, along with the comprehensiveness of system boundaries and environmental impacts assessed. Product LCA scores highly in terms of granularity (*i.e.*, as it uses product-specific, process-based data) and comprehensiveness of environmental impacts (*i.e.*, as in principle, a wide range of environmental impact categories are assessed), but is not very comprehensive in terms of system boundaries (*i.e.*, as the system boundary is narrowly constrained around a specific product with a distinct functional unit). EEIO-LCA, on the other hand, while being far more comprehensive in terms of system boundaries, lacks the granularity of product LCA. O-LCA – especially when using the novel methodological approach we put forward in this article – creates an opportunity to overcome (at least to some extent) the present trade-off between comprehensiveness (particularly of system boundaries) and granularity. As in principle the LCIA phase of O-LCA is essentially the same as in product LCA (UNEP 2015), O-LCA also – compared to organizational carbon footprinting – is more comprehensive with respect to the types of environmental impacts assessed, thus providing results beyond those associated with GHG emissions. Therefore, O-LCA creates an opportunity for a more complete balance of benefits as an analytical approach for environmental sustainability.

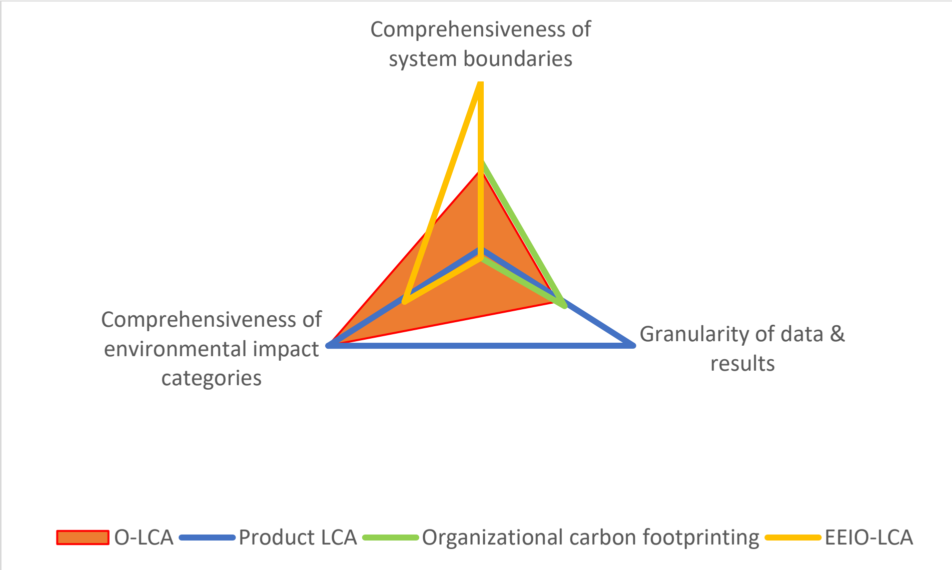

Figure 6: Conceptual comparison of product LCA, O-LCA, organizational carbon footprinting, and EEIO-LCA.

Unfortunately, however, there is very limited academic literature on O-LCA. A search of the Scopus database (on August 9, 2021, using the search string "'organizational life cycle assessment' OR 'organisational life cycle assessment' OR olca OR o-lca") yielded only 60 results (see Appendix B). Omitting irrelevant uses of the acronyms "OLCA" and "O-LCA" (*e.g.*, "ostia of the left coronary artery," "overall local class-specific accuracy," "origin of the left colic artery," "Ohio Lactation Consultant Association," "opportunistic and location-based collaboration architecture," and input-output LCA) yielded only 21 results, the majority of which (at least 13) having authorship within the same research group – in close connection with the O-LCA Guidance. The literature on "life-LCA" is even more scant, with only one seminal article on the general methodology (Goermer et al., 2020) followed by a single related case study (Bossek et al., 2021).

Moreover, as we bring forth in this article, the current state of the art – the O-LCA Guidance developed through the Life Cycle Initiative (UNEP 2015) – has methodological shortcomings that have not been addressed in the existing literature (including in the recent piece by Martínez-Blanco et al. (2020), which reflects upon the results of the O-LCA "road testing" project). In this article, therefore, we provide a constructive critique of the O-LCA Guidance, focusing on the LCA phases of goal and scope definition, life cycle inventory (LCI), and interpretation. To address our biggest criticism – concerning the flawed treatment of uncertainty in O-LCA (wherein small, non-randomized sets of "representative" products are used to model broad and diverse product portfolios) – we propose to use basic statistical sampling and inference techniques to strike a better balance of scientific rigor and practical feasibility in O-LCA. We elaborate on our novel methodology in this article, and we refer to a forthcoming contribution in which we test and demonstrate our approach on an O-LCA case study of a Canadian hospital – which provides results of interest for a critical and complex service sector with sizable, but understudied, environmental impacts (Cimprich et al., 2019; Lenzen et al., 2020; Pichler et al., 2019). In that case study, we compile new LCI data for approximately 200 goods and services used in healthcare. Our work thus aligns strongly with the three criteria given by Lifset (2013) for "raising the bar" in LCA case studies: (1) bringing new data into the literature, (2) employing a novel methodology, and (3) shedding light on a specific topic or problem of interest or significance.

3.3 Goal and scope definition

As previously discussed by Martínez-Blanco et al. (2020, 2015a), the biggest differences between product LCA and O-LCA are centered around the goal and scope definition – the first phase of any LCA, which guides the subsequent phases of LCI, life cycle impact assessment (LCIA), and interpretation. Here, as outlined in Table 1 and discussed in subsequent paragraphs, we focus on a central concept in O-LCA goal and scope definition – what is termed by ISO 14072 as the "reporting unit" – which is subject to subtle, but important, differences in terminology between the O-LCA Guidance and the international standards on which the Guidance is based (*i.e.*, ISO 14040, 14044, and 14072).

Table 1: Definitions of the "functional unit" (*i.e.*, in product LCA) and "reporting unit" (*i.e.*, in organizational LCA)

(Product) LCA terminology	**Organizational LCA equivalent**		
ISO 14044:2006	**ISO 14072:2014**	**O-LCA Guidance (UNEP, 2015)**	**Authors' interpretation**
Functional unit, defined as "quantified performance of a product system for use as a reference unit" (sec. 3.20)	*Reporting unit*, defined as "quantified performance expression of the organization under study to be used as a reference" (sec. 3.2)	*Reporting unit*, comprising the "reporting organization," the "reporting flow," and the "reference period"	*Reporting unit*, defined as the operations of the reporting organization during the reference period
Reference flow, defined as "[a] measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit" (sec. 3.29)	Not applicable	*Reporting flow*, defined as "a measure of the outputs of the reporting organization" (sec. 3.2.2)	Not applicable

The reporting unit serves as the O-LCA equivalent to the functional unit in product LCA. In its interpretation of ISO 14072, the O-LCA Guidance deconstructs the concept of the reporting unit into two constituent elements: the *reporting organization* and the *reporting flow* (UNEP 2015). The authors of the Guidance consider this interpretation, though introducing new terminology, to be in conformance with ISO 14072 (Martínez-Blanco et al., 2015a). Although the definition of the *reporting organization* is not trivial – especially for those with large and complex organizational structures and contractual relationships between multiple legal entities – our primary concern here is on the notion of the *reporting flow*.

The reporting flow is intended to replace the *reference flow* in product LCA (Martínez-Blanco et al., 2020, 2015a) – which in turn is defined by the international LCA standards as a "measure of the *outputs* from processes in a given product system required to fulfil the function expressed by the functional unit" (ISO 14044, sec. 3.29, emphasis added). Accordingly, the O-LCA Guidance defines the reporting flow as "…a measure of the *outputs* of the reporting organization" (UNEP 2015, sec. 3.2.2, emphasis added). Beyond mere wordsmithing, we see the notion of the "reporting flow" being problematic in two related ways: (1) it is a fundamental misinterpretation of the *reporting unit* concept per ISO 14072, which does *not* have an equivalent to the reference flow in product LCA, and (2) this misinterpretation leads to flawed methodological approaches in O-LCA.

On the first point, we note that, in clause 4.3.3.3 of ISO 14072 (*i.e.*, "relating data to unit process and reporting unit"), which adapts the corresponding clause from ISO 14044, the notion of the reference flow – or its O-LCA equivalent – *is conspicuous in its absence*. The omitted text reads:

> Based on the flow chart and the flows between unit processes, the flows of all unit processes are related to the reference flow. The calculation should result in all system input and output data being referenced to the functional unit (ISO 14044, sec. 4.3.3.3).

In our view, the principal reason for this omission is that, for an organization, it is typically not the case that there is a single product output, with a distinct functional unit and a corresponding reference flow. Rather, an organization – like the Brazilian cosmetics manufacturer Natura & Co., which was among the 12 O-LCA "road testing" organizations – may produce *thousands* of unique products. To be precise, Natura & Co. produces 2,600 different cosmetics and personal care products (de Camargo et al., 2019; UN Environment, 2017).

It is also notable that the word "functional" has been dropped from the O-LCA equivalent to the functional unit in product LCA. In our view, careful reading of ISO 14072 suggests that the *reporting unit* is not intended to reflect the "function" of the organization *per se* (as this notion does not make sense in the same way as in product LCA), but rather is intended to represent a *unit of operation* – the operations of the studied organization during a specified time period (termed the *reference period*). Accordingly, we challenge the validity and practicality of the "reporting flow" concept as articulated in the O-LCA Guidance (noting again that the term "reporting flow" is not used by ISO 14072). We instead suggest that the *reporting unit* (the actual term used by ISO 14072) can be broadly defined as the operations of the reporting organization during the reference period of the O-LCA. The notion of the "reference period" suggests that, in terms of the unit of analysis, O-LCA bears more similarity to corporate reporting frameworks (particularly the *GHG Protocol Corporate Standard* (WRI & WBCSD, 2004)) than to product LCA.

On the second point – regarding the consequences of what we see as a fundamental misinterpretation of the *reporting unit* concept – we highlight the flawed methodological approaches that follow. One is the loose conceptualization of the "reporting flow" for service-sector organizations (*e.g.*, in the financial sector or the healthcare sector) that do not produce physical goods. Where it is difficult to define the reporting flow in physical units of output (*e.g.*, mass, volume, or units of product), the O-LCA Guidance suggests – without a clear rationale – to define the reporting flow "…in non-physical terms, such as economic revenue and number of employees" (UNEP 2015, sec. 3.2.2). This suggestion is inconsistent with the notion of the reporting flow as "…a measure of the *outputs* of the reporting organization" (UNEP 2015, sec. 3.2.2, emphasis added). Revenue is, by definition, a financial *inflow* to the reporting organization. It could be argued that the goods and services sold (*i.e.*, the organization's outputs) generate corresponding revenues, but as elsewhere acknowledged in the O-LCA Guidance, "…spending and revenue do not correlate well with environmental impacts…" (pp. 61-62). "Number of employees" is not a *flow* at all; it is more accurately considered a *stock* (though "number of *new* employees" could be considered a flow).

Further, a bigger problem related to the "reporting flow" concept arises from the use of small, non-randomized sets of product LCAs to model the environmental impacts of large and diverse product "portfolios." The O-LCA Guidance loosely suggests defining "representative" products within an organization's portfolio – *e.g.*, one "representative" product for each product "cluster." This idea, based on the approach used by Milà i Canals et al. (2011) in their organizational carbon footprint study of Knorr foods (a subsidiary of Unilever) prior to the publication of ISO 14072 and the O-LCA Guidance, is further exemplified by the O-LCA "road testing" case study of Natura & Co. (de Camargo et al., 2019; UN Environment, 2017) we mentioned earlier. Natura

& Co.'s portfolio of 2,600 cosmetics and personal care products were grouped into 10 categories (beard, hair, body, deodorants, makeup, body oil, perfumery, sun protection, face care, and soap), and the "bestseller" in each category was used as the "representative" product for that category (de Camargo et al., 2019; UN Environment, 2017). Natura & Co.'s total environmental impacts (for the year 2013) were modelled by combining the impacts of "corporate activities" (*i.e.*, the operations of corporate headquarters and manufacturing facilities, along with product distribution and transportation) with the impacts of the 10 "bestselling" products (*i.e.*, by taking the impacts of the bestseller in each category multiplied by the total number of products sold in that category).

This "representative product" approach is scientifically invalid. First, even if revenue share can be considered a reasonable proxy for environmental intensity – an assumption rejected in the O-LCA Guidance itself (pp. 61-62) – the "bestseller" in a product category does not necessarily account for the *majority* of revenue in that category. In fact, de Camargo et al. (2019) acknowledge that the 10 "bestselling" products assessed account for only 4.3% of Natura & Co.'s total sales in the reference period. Moreover, by *explicitly selecting* "representative" products based on known characteristics (*i.e.*, sales volumes in the Natura & Co. case study, though other characteristics, such as product mass, could also be used), it creates a biased sample *by design*. A biased sample is, by definition, the *opposite* of a representative sample. Sampling design is crucial here, as the problem to be addressed – given the practical infeasibility of conducting LCAs for *all* 2,600 products – is fundamentally an inference problem (*i.e.*, inferring the environmental impacts of all 2,600 products from a comparatively small, but to some extent *representative*, sample from this "population" of products). The sampling design suggested in the current O-LCA Guidance, as applied in the Natura & Co. case study, does not allow any valid inference beyond the set of products modelled; the Natura & Co. study, effectively, is more accurately described as a compilation of 10 product LCAs (plus an LCA of Natura & Co.'s "corporate activities") rather than an *organizational* LCA.

As we will elaborate in subsequent sections – on the LCI and interpretation phases of O-LCA – we argue that a better approach would be to apply basic statistical sampling and inference techniques, which provide a well-established and scientifically rigorous approach to problems like the one encountered in O-LCA (*i.e.*, modelling broad and diverse "populations" of products – not only the *product outputs* produced by an organization, but also the *product inputs* purchased or acquired by the organization). We previously demonstrated the application of basic statistical approaches in our scoping review of the literature on sustainability in the healthcare sector (Cimprich et al., 2019), in which the purpose of this approach was to analyze the distribution of over 1,700 articles in relation to an industrial ecology framework we developed for coding the contents of the articles (*i.e.*, the percentages of articles coded under various categories defined by our framework). As we explained in our review article, the value of statistical approaches is in how they enable *inference* about a population – with a quantifiable degree of uncertainty – based on a relatively small sample from that population. Though the context is different, the fundamental methodological problem we have elucidated in O-LCA is the same as that in our healthcare sustainability scoping review: there is a *population* about which we aim to make an *inference* based on a *sample from* that population.

3.4 Life cycle inventory

The O-LCA Guidance conceptualizes two approaches to the LCI phase of O-LCA – termed "bottom-up" and "top-down" (UNEP 2015). In the "bottom-up" approach, product LCAs for the reporting organization's product "portfolio" (*i.e.*, the product outputs of the reporting organization) are scaled to the reporting organization's total output (*i.e.*, the "reporting flow") and combined with data on "supporting activities" (defined by the O-LCA Guidance as "…activities and operations of the organization that are not directly involved with the production of the products, but represent, for instance, managerial, marketing, design or R&D departments, which are key for an efficient and profitable operation of the organization" (UNEP 2015, p. 58)). The "top-down" approach "…considers the reporting organization as a whole, and adds upstream (cradle-to-gate) models for all inputs of the organization and downstream (gate-to-grave) models for all outputs" (UNEP 2015, p. 65). The bottom-up approach is also known as the "product-oriented" approach, whereas the top-down approach is also known as the "inventory-oriented" approach. Given that the terms "top-down" and "bottom-up" are commonly used to refer to EEIO-LCA and process-based LCA respectively (see, *e.g.*, Castellani et al. (2019) and Tukker & Jansen (2006)), and therefore the different use of these terms in the O-LCA Guidance can be confusing, the alternative terms "product-oriented" and "inventory-oriented" might be better – although the latter term is somewhat awkward given that *any* LCA involves a life cycle *inventory*. A "hybrid" approach is exemplified by the Natura & Co. case study discussed previously – where we criticized the use of a small set of "representative" products to model the company's broad and diverse product portfolio.

Following broader convention in LCA practice (Ecoinvent, 2021b; European Commission, 2010; Life Cycle Initiative, 2021; US EPA, 2006) – we prefer to use the terms *foreground data* and *background data*. Foreground data pertain to the foreground processes (*i.e.*, in the operations of the reporting organization during the reference period); these data describe and quantify the inputs to, and outputs from, the organization's operations (*e.g.*, energy and water used, wastes generated, and goods and services purchased or acquired). Background data pertain to background processes (*i.e.*, the upstream processes associated with the production of goods and services purchased or acquired by the reporting organization, and the downstream processes associated with the use and end-of-life of goods and services produced by the reporting organization). Foreground and background data need to be "matched" to derive the LCI and LCIA results.

While high-quality foreground data are likely to be readily available in O-LCAs (as foreground data are, by definition, specific to the organization's own operations, and these data are already needed by the organization for other purposes, such as inventory management and financial accounting), background data pose a fundamental methodological and practical challenge that arises from the scale and complexity of modelling background processes in O-LCA. In the previous section – on O-LCA goal and scope definition – we noted that, as demonstrated through the Natura & Co. case study (de Camargo et al., 2019; UN Environment, 2017), an organization's product "portfolio" (*i.e.*, the product *outputs from* the reporting organization) can encompass *thousands* of unique goods and services. The same is true of organizational supply-chains (*i.e.*, the product *inputs to* the reporting organization). Recognizing this practical reality, the O-LCA Guidance provides a set of criteria for prioritizing LCI data collection efforts; these criteria include the common quantitative LCA "cut-off" criteria of environmental significance, mass or energy values, and economic values – along with the largely qualitative "organizational aspects" of *suppliers' closeness*, *influence*, *risk*, *stakeholders*, *outsourcing*, and *sector guidance* (UNEP 2015, p. 60).

All these criteria are problematic. Applying a cut-off based on environmental significance would require pre-existing knowledge, or at least well-educated assumptions, of the potential environmental impacts of all foreground and background processes. Applying a cut-off based on mass, energy, or monetary values assumes that these variables are strongly associated with, and therefore good predictors of, potential environmental impacts – an assumption that would also need to be supported by evidence. As acknowledged in the O-LCA Guidance, "…there is no theoretical or empirical basis that guarantees that a small mass or energy contribution will always result in negligible environmental impacts […] as with energy and mass basis, spending and revenue do not correlate well with environmental impacts, and so this criterion should not be used alone" (pp. 61-62). Moreover, using mass, energy, or monetary values for cut-offs requires that these quantities are *known* for all inputs and outputs in the O-LCA system boundary. While organizations may have comprehensive financial data on their inputs and outputs (*e.g.*, for management and accounting purposes), this may not be the case for mass or energy (*e.g.*, it is unlikely that *every* organization is going to precisely weigh the mass of *every* product it receives, and it is unlikely to have such information for *every* product from *every* supplier).

Collecting data only for the closest suppliers (*i.e.*, at the top levels of the supply-chain) is convenient, but probably not representative; this approach could overlook the most important upstream processes. The idea of prioritizing data collection based on the reporting organization's degree of "influence" is quite ambiguous. For example, if an organization purchases products that are produced in a relatively environmentally impactful way, it could switch to another supplier that produces similar products in a less environmentally impactful way. In other words, though it cannot directly control its suppliers' operations, it can influence the environmental impacts of its supply-chains through sourcing and purchasing decisions. Another example is electricity use; even if the upstream processes of electricity supply are beyond the immediate control of the reporting organization, the reporting organization can reduce its environmental impacts by reducing its electricity use (*i.e.*, through improved energy efficiency in its own operations) and/or by using alternative sources of electricity (*e.g.*, by installing its own renewable electricity generation capacity). Prioritizing data collection based on the reporting organization's risk exposure would, as with environmental, mass, energy, or monetary values, require substantial existing evidence; indeed, as elsewhere suggested by the O-LCA Guidance, risk assessment could be one of the organization's motivations for doing an O-LCA in the first place. Stakeholder concerns and/or sector guidance could, and probably should, be considered in an O-LCA, but they can be a source of bias and should not in our view be the sole basis for scoping or other methodological decisions.

Another option, as recognized by the O-LCA Guidance (box 6, p. 62), is to use an EEIO-LCA model. In essence, the EEIO-LCA approach combines regional or multi-regional economic input-output tables – containing data on aggregated economic transactions (*i.e.*, exchanges of goods and services, in monetary terms) between industry sectors – with data on the emissions intensity of each sector. This approach is widely used to assess the environmental impacts of large-scale systems, and in what is commonly known as the "hybrid" LCA approach, is used to fill data gaps in process-based LCAs. EEIO-LCA has several limitations, however. Uncertainties attached to EEIO-LCA and hybrid LCA results, if quantified at all, can be significant (Berners-Lee et al., 2011; Jakobs et al., 2021). The relatively infrequent updates of input-output tables result in time lags that limit the temporal representativeness of EEIO-LCA results; *e.g.*, at the time of writing, Exiobase 3 dates to at least 2011 (EXIOBASE Consortium, 2022), the United States (US) EEIO model dates to at least 2012 (US EPA, 2022), and the Open-IO Canada model dates to 2009 (CIRAIG, 2022). Perhaps most importantly, the high level of aggregation in EEIO-LCA creates the potential for mismatching of sectoral categories between expenditure amounts (*i.e.*, as the foreground data in an O-LCA) and EEIO models (*i.e.*, as the source of background data), and limits the capacity of the EEIO-LCA approach to provide actionable information to guide decision-making in organizations. Moreover, according to ISO 14072:

> …even for an OLCA, *a product perspective (e.g. purchased products) shall be taken* when assessing the supply chain. To do so, *product level data should be used* and this represents the interface to the domain of LCA as defined in ISO 14040 and ISO 14044. As a consequence, the (theoretical) advantage of OLCA of not having to cope with numerous product life cycles might not apply anymore. *From a conceptual perspective, there is no OLCA without a product perspective* (sec. 5.3, emphasis added).

Given the shortcomings of existing approaches to modelling background processes in O-LCA, we propose a simple, but novel, approach to strike a better balance of scientific rigor and practical feasibility in O-LCA. Our proposal is to use basic statistical sampling and inference techniques – specifically, random sampling (also known as *probability sampling*) and confidence intervals for the mean (*i.e.*, average) environmental intensity, per unit of expenditure, of product *inputs* (noting that a similar approach could also be applied for modelling the use and end-of-life of product *outputs*). The estimated average environmental intensity (technically, the *weighted* average, where the weights are the corresponding cost shares of the products, should be used – as mathematically demonstrated in Appendix B) can then be multiplied by the total expenditure to estimate the total LCIA results. This statistical approach may not need to be applied to all product inputs (*e.g.*, energy and water use could be considered separately); to the extent that LCI data of acceptable quality are readily available, those data should be used.

3.5 Life cycle interpretation

Regarding the interpretation phase of O-LCA, we focus on the critical methodological aspects of data quality and uncertainty – which, as demonstrated through existing case studies like that of Natura & Co. (de Camargo et al., 2019; UN Environment, 2017) discussed earlier, are poorly addressed in current O-LCA practice. The O-LCA Guidance presumes that these aspects are essentially the same as in product LCA, and therefore devotes only five pages (three in the LCI section and two in the interpretation section) to this topic (UNEP 2015). The Guidance points to common LCI data quality criteria (*i.e.*, temporal representativeness, geographical representativeness, technological representativeness, precision, completeness, reproducibility, and reliability). As in product LCA, these criteria need to be used to assess the quality of foreground data, background data, and the links between foreground and background data. We argue, however, that O-LCA presents greater methodological and practical challenges due to the tendency of the studied system to be orders of magnitude greater in scale and complexity.

To help overcome these challenges, we have proposed to use basic statistical sampling and inference techniques – *i.e.*, estimating the mean (or average) environmental intensity of the reporting organization's product inputs (or product outputs) based on a random sample from this "population" of products. Details on how to construct confidence intervals for the LCIA results, and how to estimate the minimum sample size required for a targeted margin of error at a specified confidence level, with a hypothetical worked example, are given in Appendix B. Using the example in Appendix B, if the estimated average environmental intensity (of the 1,500 unique products purchased) is 0.3333 kg CO_2 eq./$, and the margin of error (based on a sample of 38 unique products) is 0.1667 kg CO_2 eq./$, it could be concluded that, *with 95% confidence*, the total global warming potential (*i.e.*, for the total expenditure of $1,500,000 on 1,500 unique products) is between 250 and 750 t CO_2 eq. Notably, this margin of error accounts for only the uncertainty attributable to random sampling; it does not account for the uncertainty attributable to the quality of the underlying LCI data used to model the products in the sample – something that is worthy of further consideration but is beyond the scope of this article.

We recognize that the use of economic values in LCA – particularly in relation to allocation or "multi-functionality" problems – is controversial (see, *e.g.*, Pelletier et al. (2015), Pelletier & Tyedmers (2011, 2012), and Weinzettel (2012)). However, the fundamental problem that motivates our approach is *not* an allocation problem – wherein "allocation" is defined as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (ISO 14044, sec. 3.17). Rather, the problem we confront is an *inference* problem; there is a *population* (*i.e.*, the full set of product inputs to the reporting organization), which has an unknown *population parameter* (*i.e.*, average environmental intensity, and ultimately total environmental impacts) that we estimate by *sampling from* this population. *Random* sampling – in contrast to the sampling approaches currently used in O-LCA – ensures that the sample will be free of bias (*i.e.*, there is no systematic under- or over-representation of low- or high-intensity products), thereby enabling statistical inference.

Nor do we consider this to be a "cut-off" approach – defined by ISO 14044 as "…specification of the amount of material or energy flow or the level of environmental significance associated with unit processes or product system to be excluded from a study" (sec. 3.18). Truly random sampling, by definition, means that *each* of the reporting organization's product inputs have an equal probability of being included in the sample. Moreover, we find no requirements of ISO 14044 or 14072 that preclude this approach. Regarding the former (ISO 14072, sec. 4.2.3.3.1), our approach does not significantly change the overall conclusions of an O-LCA (depending on the definition of "significance" – particularly in terms of the level of precision and confidence). The reasons (*i.e.*, the practical infeasibility of compiling LCI data for potentially thousands of products) and implications (*i.e.*, sampling error, quantified in the form of confidence intervals) for omissions – if they can be considered as such – are clear.

Further, our approach – despite the use of economic values – should not be confused with EEIO-LCA; the sole purpose of using economic values in our approach is to express the environmental intensities of different products in common units (*e.g.*, kg CO_2 eq./$) as necessary to calculate an *average* intensity value that serves as the population parameter to be estimated. As mathematically demonstrated in Appendix B, the calculated environmental impacts for a given set of products – based on the *weighted* average environmental intensity of the products – is the same irrespective of how the "weighting" is done (*e.g.*, by each product's share of the total mass or total cost of all products purchased). Regardless of which "weighting" is used, the result is mathematically equivalent to the sum of the total environmental impacts for the number of units of each product purchased (*i.e.*, environmental impact of product A × number of units of product A + environmental impact of product B × number of units of product B, and so on). Therefore, provided the *weighted* average is used, there is no *theoretical* reason to prefer physical values (*e.g.*, product mass) over economic ones – while practicality will tend to favour the latter (*i.e.*, as it is much easier to acquire data on product costs than, *e.g.*, product masses, and mass may not be a suitable unit of account for all products – *e.g.*, for energy supply and intangible services). For greater clarity, we reiterate that this is *not* an “allocation” approach in the conventional sense of this term in LCA; we are not dismissing the theoretical and practical arguments (see, *e.g.*, Pelletier & Tyedmers (2011, 2012), and Weinzettel (2012)) against the use of economic values for that purpose. We also note that our use of the term “weighting” in this sense is not to be confused with the specific meaning of the term “weighting” in LCIA (*i.e.*, weighting multiple environmental impact categories to derive a “single score” of environmental performance).

As can be seen in the equations in Appendix B, the margin of error is highly sensitive to the sample standard deviation and the *sample* size, but *not* to the *population* size (unless the sample is more than about 10% of the population, following the rule of thumb suggested by DeVeaux et al. (2015)). In our hypothetical example, the “finite population correction” is not necessary, so the population size is irrelevant to the margin of error. All other things being equal, the margin of error would be the same if there were 1.5 *million* unique products purchased. This counter-intuitive property of statistical inference – that it is the *sample* size that matters, irrespective of the *population* size – suggests that our approach has potential to be highly scalable (*i.e.*, to O-LCAs of even the largest organizations). However, there is a trade-off between precision and confidence; all other things being equal, greater precision (*i.e.*, a smaller margin of error) requires lower confidence, and vice versa. Further, a very heterogeneous population – with a high standard deviation of environmental intensity – will require a larger sample for a specified level of precision and confidence. Increasing the sample size will improve both precision and confidence, but it will also require more work. Organizations therefore need to balance the competing objectives of confidence, precision, and practicality.

3.6 Conclusion

Despite our criticisms of the Life Cycle Initiative's Guidance on O-LCA, and current O-LCA practice following the Guidance, we remain supportive of O-LCA. By scaling the LCA framework while retaining its core strengths, O-LCA has the potential to provide an even more powerful decision support tool for sustainable development – one that provides a compelling balance of granularity (*i.e.*, of data and results), and comprehensiveness (*i.e.*, of system boundaries and of the types of environmental impacts assessed). Our core contribution in this article is a *constructive* critique of the current state of the art, with concrete proposals – particularly regarding the use of basic statistical sampling and inference techniques – to improve O-LCA from a methodological and practical standpoint.

Our next step will be to test and demonstrate our proposals on O-LCA case studies. Specifically, in a forthcoming contribution, we will present our novel O-LCA of a Canadian hospital – which provides results of interest for a critical and complex service sector with sizable, but understudied, environmental impacts (Cimprich et al., 2019; Lenzen et al., 2020; Pichler et al., 2019). To allow a comprehensive and detailed assessment of the hospital's environmental impacts, we accessed – subject to confidentiality – a wealth of foreground data covering energy and water use, solid waste generation, procurement (including purchases of 2,927 unique products used in the hospital), and outsourced services (including patient transportation, housekeeping, laundry, and food services). Using basic statistical sampling and inference techniques – as we have proposed in this article – the foreground data are matched to new background data we compiled for approximately 200 goods and services used in healthcare. Our work thus aligns strongly with the criteria given by Lifset (2013) for "raising the bar" in LCA case studies.

Supplementary content

See Appendix B.

Data Availability Statement

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

Acknowledgements, funding, and conflicts of interest

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Chapter 4

4. Organizational LCA of a hospital

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After the submission to the journal, and the defence of this thesis, some revisions have been made in this Chapter to further improve clarity, transparency, and overall scholarly quality. The revisions do not impact the core methodology, findings, or contribution of the manuscript.

4.1 Summary

We present our organizational life cycle assessment (O-LCA) of a 40-bed hospital in British Columbia, Canada – to our knowledge, the first O-LCA in the healthcare sector. Healthcare is a critical and complex service sector with a sizable "environmental footprint" – with greenhouse gas (GHG) emissions amounting to 5-10% of the national total in developed economies like Canada and the USA. But recent reviews indicate that the existing literature on "healthcare sustainability" is fragmented, and the evidence base – to inform policy, management, and clinical practice – is weak. Drawing from our hospital O-LCA, we make four broad observations about the scale, complexity, and variability of the environmental footprint of healthcare facilities. First, in terms of scale, the total GHG emissions from our studied hospital – from one year of operations – are 3,500-5,000 t CO_2 eq., with 95% confidence. Second, energy and water use account for a major "hotspot" in the hospital's environmental footprint, despite the hospital being in a region with a low-carbon electrical grid. Third, transportation (of patients and of goods) and waste management – despite being popular and visible focus areas for healthcare sustainability – contribute comparatively little to the hospital's total footprint. Finally, a complex set of hotspots

is attributable to less visible aspects of the hospital's operations – particularly the production of the 2,927 unique products purchased through the hospital's supply-chains. In the process of conducting our hospital O-LCA, we also compiled new LCA data for approximately 200 goods and services used in healthcare.

4.2 Introduction

Healthcare is a critical and complex service sector that has only recently begun to come to terms with its sizable "environmental footprint"[7] (Lenzen et al., 2020; Pichler et al., 2019) – with greenhouse gas (GHG) emissions estimated at 5-10% of the national total in developed economies like Canada (Eckelman et al., 2018) and the USA (Eckelman et al., 2020; Eckelman & Sherman, 2016). These estimates, calculated using environmentally extended input-output life cycle assessment (EEIO-LCA; in what is commonly referred to as a "top-down" approach), account for both the direct emissions from healthcare facilities (*e.g.*, from on-site fuel combustion and from releases of anesthetic gases – which are potent GHGs (Sherman et al., 2012)) – and the indirect emissions attributable to the wide range of purchased goods and services (*e.g.*, pharmaceuticals and other medical products, along with laundry, housekeeping, and food services) used in healthcare. As in all other sectors, the environmental footprint *of healthcare* can be linked to public health harms; in the USA, these harms – measured in

[7] The term "environmental footprint" is a common shorthand (see, *e.g.*, Daughton & Ruhoy (2009), Ivanova et al. (2016), Jiménez-González et al. (2004), Lim et al. (2013), Panyakaew & Fotios (2011), Sáez-Almendros et al. (2013), Santero et al. (2011), and Talibi et al. (2022)) for the more technically precise terminology, codified in international standards (ISO 14040:2006, ISO 14044:2006) used in environmental life cycle assessment (LCA), particularly the terms "emissions" (*e.g.*, greenhouse gas emissions, other air emissions, emissions to water, and emissions to soil) and "potential environmental impacts" (*e.g.*, global warming potential, acid precipitation potential, smog formation potential, and eutrophication potential). A related term, "carbon footprint," is widely used to refer to greenhouse gas emissions (Finkbeiner, 2009), typically expressed in units of global warming potential (*i.e.*, as a mass of "carbon dioxide (chemical formula CO_2) equivalent" emissions).

disability-adjusted life-years (DALYs) lost – were found to be of similar magnitude to the harms attributed to preventable medical errors (Eckelman et al., 2020; Eckelman & Sherman, 2016).

Though it provides a high-level assessment of the magnitude of healthcare's environmental footprint, the "top-down" approach has several limitations. First, the uncertainties attached to the EEIO-LCA results, if quantified at all, are significant. A top-down study of the Australian healthcare sector, published in *The Lancet Planetary Health*, estimated a carbon footprint of 25-46 Mt CO_2 eq. in 2014-15 (Malik et al., 2018). In another top-down study, also published in *The Lancet Planetary Health*, the carbon footprint of the Chinese healthcare sector was estimated at 267-363 Mt CO_2 eq. in 2012 (Wu, 2019). Crucially, these estimates are calculated at the 68% confidence level – as opposed to the 95% confidence level commonly used in most scientific disciplines. Most recently, in an assessment of the carbon footprint of the National Health Service (NHS) in England (Tennison et al., 2021) – also published in *The Lancet Planetary Health* – the authors acknowledge that "[d]ue to the hybrid approach combining multiple bottom-up datasets with top-down […] results, it was not feasible to conduct a comprehensive uncertainty analysis" (p. e87). In their top-down study of the Canadian healthcare sector, published in *PLOS Medicine*, Eckelman et al. (2018) estimate the public health harms attributable to the sector's environmental footprint at 4,500-610,000 DALYs lost annually. This very wide range is attributable to the uncertainty of modelling the public health consequences of GHGs and other environmental emissions; it does not reflect the uncertainty attributable to the quality of the underlying data used to estimate the quantities of these emissions (Eckelman et al., 2018).

Another, related weakness of the top-down approach arises from the infrequent updates of national input-output tables, resulting in time lags that limit the temporal representativeness of

EEIO-LCA results. In the Australian study cited previously, the input-output tables were from 2014-15 (Malik et al., 2018). In the Chinese study, the 2012 national input-output table was the most recent available (Wu, 2019). In the NHS England study, data from the UK input-output tables (for the top-down portion of the study) for 1997-2016 were used to represent the study period of 1990-2019 (Tennison et al., 2021). In the Canadian study, the top-down data are for 2009-2015 (Eckelman et al., 2018).

Arguably the most important limitation of the top-down approach, however, is its high level of aggregation. Input-output tables contain data on aggregated economic transactions between industry sectors, as opposed to a detailed "bottom-up" accounting of the purchase and sale of specific goods and services. In the Canadian healthcare study (Eckelman et al., 2018), for example, the National Health Expenditures (NHEX) database, maintained by the Canadian Institute for Health Information, was used in combination with the Open IO-Canada database developed by the International Reference Centre for the Life Cycle of Products, Processes and Services (CIRAIG) at École Polytechnique de Montréal. In the NHEX database, health expenditures are classified into 13 broad categories, *e.g.*, *hospitals*, *physicians*, *prescribed drugs*, and *non-prescribed drugs* (Eckelman et al., 2018). The health expenditure data from the NHEX database – serving as what are commonly referred to as *foreground data* (*i.e.*, data that describe and quantify the inputs to, and outputs from, the healthcare sector) in LCA practice (Ecoinvent, 2021b; European Commission, 2010; Life Cycle Initiative, 2021; US EPA, 2006) – were matched to *background data* from the Open IO-Canada database (*i.e.*, data on the environmental emissions attributable to the production, use, and/or end-of-life treatment of the inputs and outputs). The matching (or *mis*matching) between the foreground and background data is another critical methodological aspect and a significant potential source of error – a concern compounded by the limited granularity in the categorization of inputs and outputs in top-down datasets. For example, the NHEX categories of *vision care services*, *capital*, and *public health* were matched with the "commodity codes" of *other health practitioner services*, *non-residential building construction*, and *other health and social assistance services*, respectively, from the Open IO-Canada database (Eckelman et al., 2018). Moreover, although the authors used the most recent NHEX data (*i.e.*, for 2015) available at the time of the study (*i.e.*, as the foreground data), the Open IO-Canada database (*i.e.*, as the source of background data) was, and still is, based on Canadian input-output tables from 2009 (CIRAIG, 2022).

Along with the potential for *mis*matching between foreground and background data, the high level of aggregation in the top-down approach yields a low level of resolution in the resulting environmental footprint estimates. In the NHS England study, 62% of GHG emissions (in 2019) were attributed to the NHS "supply chain" – within which the largest contributions were from the broad categories of *pharmaceuticals and chemicals*, *medical equipment*, and *business services* (Tennison et al., 2021). In the Canadian study (Eckelman et al., 2018), the largest contribution (24%) to the total GHG emissions from the healthcare sector was attributed to a category labelled as "other." To provide more actionable information to support decision-making in healthcare facilities, *bottom-up* environmental footprinting studies – following a process-based LCA approach that uses more granular foreground and background data – are needed. The authors of top-down studies acknowledge this point. In the NHS England study, the authors note that their work "…underscores […] the need for investing in bottom-up data collection through robust and validated information systems to increase the accuracy and resolution of emissions accounting and inform focused interventions" (Tennison et al., 2021). In the Canadian study, the authors acknowledge in the abstract that "[a] limitation of this national-level study is the use of aggregated data and multiple modeling steps to link healthcare expenditures to emissions to health damages. While informative on a national level, the applicability of these findings to guide decision-making at individual institutions is limited" (Eckelman et al., 2018).

Moreover, our comprehensive scoping review, based on a representative sample from over 1,700 articles published between 1987 and 2017, indicates that the literature on "healthcare sustainability" is fragmented, and the evidence base – to inform healthcare policy, management, and clinical practice – is weak (Cimprich et al., 2019). Aside from top-down studies, like those previously cited, applications of environmental footprinting have been mostly limited to a growing, albeit sporadic, set of bottom-up LCA studies (or carbon footprint studies) of specific medical products and procedures, such as medical imaging (see, *e.g.*, Cimprich et al. (2018) and Esmaeili et al. (2015)), birthing procedures (Campion et al., 2012), surgeries (see, *e.g.*, Thiel et al. (2015), Thiel et al. (2017), and Venkatesh et al. (2016)), and various medical devices and supplies (see, *e.g.*, Campion et al. (2015), Eckelman et al. (2012), Ibbotson et al. (2013), and Leiden et al. (2020)). Although these bottom-up studies have much higher resolution than top-down studies, they are also much less comprehensive in their system boundaries – typically covering only one product or procedure (or two, given that these studies are often comparative). A "hybrid" approach combining bottom-up data (typically for facility energy and water use and waste generation) with top-down data (typically for the production of materials, chemicals, instruments, devices, equipment, and other products used in healthcare facilities) is exemplified by a recent carbon footprint study of an inpatient unit and an intensive care unit (ICU) in a U.S. hospital (Prasad et al., 2022). Although this approach helps address some of the limitations of bottom-up approaches, it does not fully overcome the trade-off between the comprehensiveness of top-down approaches and the granularity of bottom-up approaches.

Further, subsequent to our healthcare sustainability scoping review, Drew et al. (2021) conducted a deeper systematic review and critical evaluation of 44 LCA-based studies in the areas of surgery and anesthesia, in which they found numerous weaknesses in the methodology of these studies (*e.g.*, failing to clearly explain and justify crucial scoping and methodological choices, failing to report numerical LCA results, failing to report results for a broad suite of environmental indicators, and failing to adequately discuss or evaluate sources of uncertainty). And although there has been a proliferation of healthcare-related LCAs in recent years (with over 50% of the 44 studies evaluated by Drew et al. (2021) having been published after 2017 – and therefore not included in our previous scoping review), the existing literature still covers only a tiny fraction of the thousands of unique products used in healthcare. In short, the current evidence base on healthcare's environmental footprint is markedly deficient in quantity *and* quality. Unfortunately, this practical reality ultimately becomes a limitation of our work in this article as well; we will return to this point in our discussion.

In this article, we present what is – to the best of our knowledge – the first application of organizational LCA (O-LCA) in the healthcare sector. O-LCA is a relatively new form of LCA that scales the basic methodological framework of product LCA to assess the environmental footprint of an organization's operations – including direct and indirect emissions – over a specified time period (Martínez-Blanco et al., 2015b). Specifically, we conduct an O-LCA of a hospital in the Canadian province of British Columbia for its 2019 fiscal year. We follow the requirements and guidelines from the main reference documents on O-LCA: ISO 14072:2014, which adapts the international standards on product LCA (*i.e.*, ISO 14040:2006 and ISO 14044:2006), and the O-LCA Guidance developed through the Life Cycle Initiative hosted by the United Nations Environment Programme (UNEP, 2015), which builds upon ISO 14072. To allow a comprehensive and primarily bottom-up assessment of the hospital's environmental footprint, we accessed – subject to confidentiality – a wealth of foreground data covering energy and water use, solid waste generation, procurement (including purchases of 2,927 unique products used in the hospital), and outsourced services (including patient transportation, housekeeping, laundry, and food services). With a novel application of basic statistical sampling and inference techniques, the foreground data are matched to new background data we compiled for approximately 200 goods and services used in healthcare. Our work thus aligns strongly with the three criteria given by Lifset (2013) for "raising the bar" in LCA case studies: (1) bringing new data into the literature, (2) employing a novel methodology, and (3) shedding light on a specific topic or problem of interest or significance.

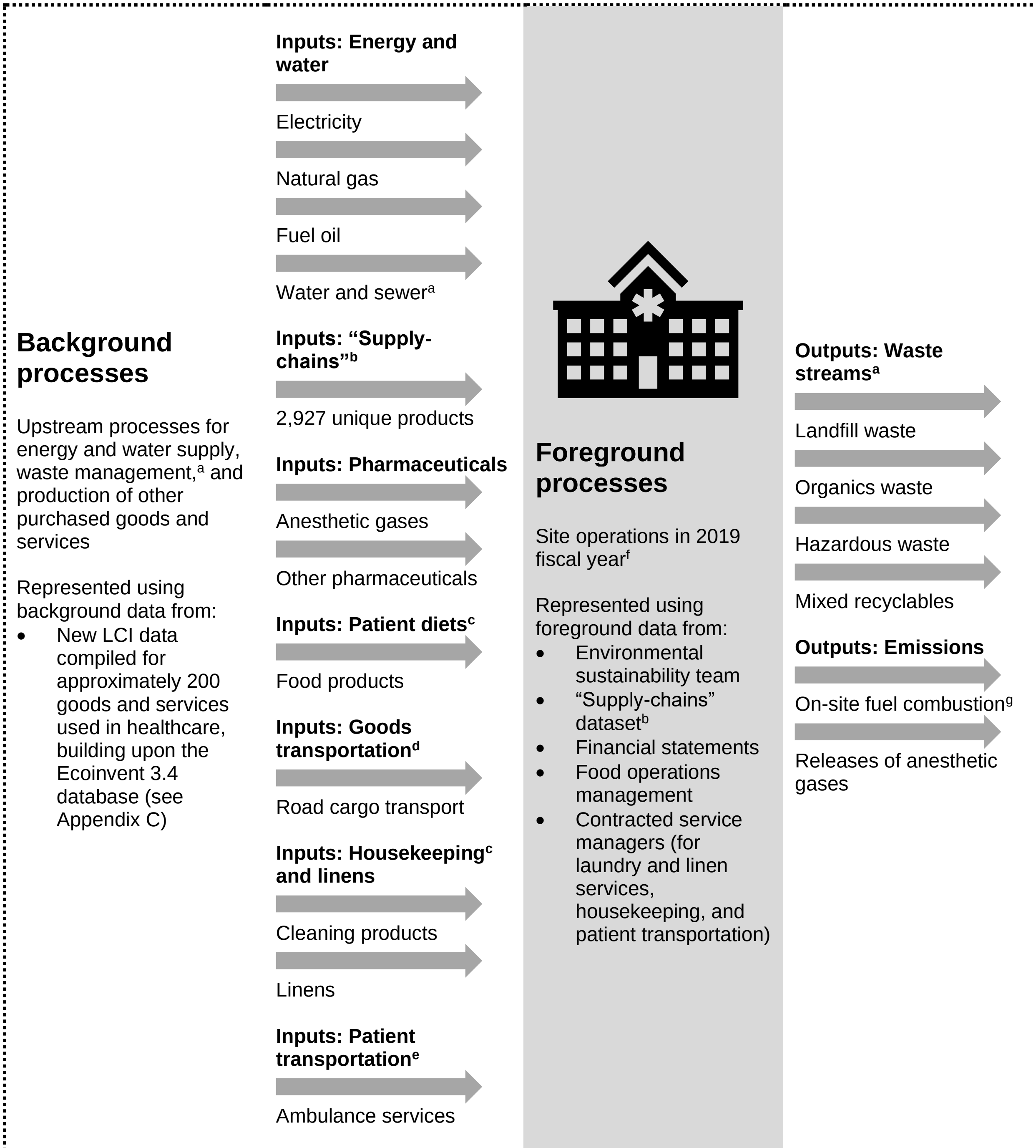
4.3 Methods

Like any LCA, an O-LCA has four connected phases – goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and life cycle interpretation (ISO 14072; UNEP 2015).

The goal of our hospital O-LCA is to comprehensively assess the environmental footprint of the hospital's operations and highlight "hotspots" (*i.e.*, major contributors to the footprint) – thereby building an evidence base to support decision-making around environmental sustainability. Following best practice in LCA, the assessment is intended to cover a range of environmental considerations beyond GHG emissions (*i.e.*, as in carbon footprinting) and thus avoid potential "problem-shifting" – a phenomenon highlighted in an editorial by Finkbeiner (2009). As the reporting organization in this O-LCA, the studied hospital is a relatively small (approximately

40-bed) and rural facility in the Canadian province of British Columbia – which has a relatively mild climate and a low-carbon electrical grid with a large share of hydroelectric generation (Canada Energy Regulator, 2021a). The hospital has also recently undergone a major renovation, with a Leadership in Energy and Environmental Design (LEED®) green building certification. These contextual factors suggest that the hospital's environmental footprint is likely to be, comparatively speaking, on the low end (*i.e.*, compared to larger and more complex hospitals, ones without green building certifications, and ones in locations with harsher climates and/or higher-carbon electrical grids), and in any case they limit the generalizability of our O-LCA results. Nonetheless, as the hospital provides a wide range of healthcare services (including outpatient services, surgical services, birthing services, emergency services, medical imaging, and laboratory services), it makes an informative case study in the understudied domain of healthcare sustainability. In conformance with ISO 14072 and UNEP (2015), our O-LCA results are not intended to be used in comparative assertions intended to be disclosed to the public.

We define the *reporting unit* – the equivalent to the functional unit in product LCA (ISO 14072; UNEP 2015) – as the operations of the hospital site, including all facilities on the premises (*i.e.*, the reporting organization), during the health authority's 2019 fiscal year (*i.e.*, the "reference period" – April 1, 2018 through March 31, 2019). As depicted in Figure 7, the O-LCA system boundary encompasses the inputs to, and outputs from, the hospital's operations during this period. The inputs include the energy and water used by the hospital, the products purchased through the hospital's "supply-chains" (2,927 unique products in total), the pharmaceuticals used in the hospital, the food products used to feed patients in the hospital, the transportation of goods to the hospital, the contracted housekeeping services provided at the hospital, the linens used in the hospital, and the ambulance services for transporting patients to the hospital. Aside from the healthcare services ultimately provided by the hospital, the hospital's outputs include its solid waste streams and its direct emissions from on-site fuel combustion and releases of anesthetic gases.



[a]Although waste streams are *physical outputs* from the hospital, the *treatment and management* of the wastes (including sewerage of wastewater) – in external facilities – is considered a *service input* in the O-LCA Guidance (UNEP, 2015) and in the software (*i.e.*, OpenLCA 1.10.2) and database (*i.e.*, Ecoinvent 3.4) used to model the treatment processes.

[b]In this article, the term "supply-chains," when put in quotation marks, refers to a specific dataset that documents a major portion of the hospital's procurement – approximately 20% of the hospital's total expenses. We say "approximately" due to the slight discrepancy in time periods between the "supply-chains" dataset (*i.e.*, calendar year 2018) and the financial statements (*i.e.*, fiscal year 2019 – April 1, 2018 through March 31, 2019). Not all

purchases and contracts are covered in this dataset, however (*e.g.*, expense accounts pertaining to the purchasing of pharmaceuticals are not covered in this dataset).

[c]The energy and water used, and wastes generated, in food, laundry, and housekeeping services are included in the totals for the site (*i.e.*, in the categories of “energy and water” and “waste streams”). Though not necessarily the case for all hospitals, in our case the hospital’s laundry services are performed on-site.

[d]Includes shipping, delivery, and courier services.

[e]To avoid double-counting (*i.e.*, as patients are transferred between sites, the transportation *from* site A is also the transportation *to* site B), only *inbound trips* (*i.e.*, trips *to* the studied hospital site) are included in this study.

[f]April 1, 2018 through March 31, 2019.

[g]Due to the structure of the Ecoinvent 3.4 database, the emissions from on-site fuel combustion are combined with the upstream emissions from the production of the fuels.

Figure 7: Organizational LCA system boundary for a hospital in British Columbia, Canada

We exclude employee travel and commuting, along with patient and visitor travel using vehicles not owned or controlled by the hospital or its contracted patient transportation (*i.e.*, ambulance) service providers. As detailed in Appendix C, various other expense accounts in the hospital’s financial statements – which, as suggested by the O-LCA Guidance (UNEP, 2015), we used to check the completeness of our study scope – are also excluded due to data gaps, presumptions of limited environmental significance (*e.g.*, for largely intangible aspects such as labour-related expenses and information technology services), and/or comparatively small expenditures in relation to the hospital’s total expenses.

As we alluded to earlier, the LCI phase of an O-LCA requires two kinds of data – which, following convention in LCA practice (Ecoinvent, 2021b; European Commission, 2010; Life Cycle Initiative, 2021; US EPA, 2006), we refer to as *foreground data* and *background data*. Foreground data pertain to the foreground processes (*i.e.*, in the operations of the hospital during the reference period); these data describe and quantify the inputs to, and outputs from, the hospital’s operations – as depicted in Figure 7. Background data pertain to background processes (*i.e.*, the upstream processes associated with the production of goods and service inputs). Foreground and background data need to be “matched” to derive the O-LCA results.

Subject to confidentiality, we obtained a wealth of foreground data for the hospital – including data from the hospital’s environmental sustainability team (*i.e.*, on the physical quantities of energy and water used, and wastes generated, in the hospital’s operations), and from the hospital’s “supply-chains” dataset, financial statements, Facilities Maintenance and Operations team, Food Operations Manager, and contracted service managers (particularly for patient transportation along with hospital housekeeping, laundry, and linen services). Further details on our foreground and background data are provided in Appendix C.

To facilitate the matching of foreground and background data, and to support the use of our background data in future O-LCAs (in healthcare and in other sectors), we classified the hospital’s inputs using the United Nations Standard Products and Services Code® (UNSPSC®) taxonomy – an internationally standardized classification of goods and services throughout the economy (see link in reference list). As detailed in Appendix C, the corresponding background process for each input is identified by a UNSPSC® commodity code and description (*e.g.*, “83101602: Supply of fuel oil”). In cases where additional granularity is needed, we addended the UNSPSC® commodity code with hyphenated digits, and we addended the commodity description with a comma followed by further descriptors (*e.g.*, “83101800-01: Electric utilities, electricity, low voltage”). The background processes – for a total of approximately 200 goods and services – are modelled using data from Ecoinvent 3.4, supplemented by a wide range of literature references, assumptions, and estimations as documented in Appendix C. Particularly with respect to background data, it is important to note that the previously highlighted limitations of existing healthcare-related LCAs – in both quantity *and* quality – ultimately become a limitation of our hospital O-LCA. We will return to this point in our discussion.

For the products in the “supply-chains” dataset – containing over 50,000 purchase records for the 2018 calendar year (noting that this time period deviates slightly from the reference period for our O-LCA) – we took a simple, but novel, approach as illustrated in Figure 8 and explained in subsequent paragraphs.

The prefix “AC” (*i.e.*, the initials of the first author) indicates that these product categories represent an aggregation of the hospital’s 218 expense accounts used for financial accounting.

The population size (*i.e.*, the total number of unique products) of each product category is represented by an uppercase *N*, and the corresponding sample size is represented by a lowercase *n*.

The notation *n*', where applicable, indicates that some products in the initial random sample were excluded, typically due to insufficient item description or lack of background LCI data (see Appendix C for further details). To be conservative (*i.e.*, to help avoid underestimation of uncertainty), the margins of error are calculated using *n'* instead of *n*, as applicable.

Although this figure shows only the results for global warming potential (GWP), the same approach was used to generate the O-LCA results for each of the environmental indicators in the TRACI impact assessment method used in this study.

Figure 8: Flow diagram of basic statistical sampling and inference approach used to model the "environmental footprint" of the hospital's "supply-chains"

We used basic statistical sampling and inference techniques – random sampling and confidence intervals – to balance scientific rigour (*i.e.*, in terms of achieving representative coverage of the wide array of products purchased) with practical feasibility (*i.e.*, to minimize the number of product LCAs needed to achieve representative coverage). Each of the 50,000+ line items are assigned by the hospital to one of 218 expense accounts for financial accounting. Recognizing that many of these line items represent the same product being purchased multiple times (*i.e.*, on different dates, by different people, *etc.*), we identified 2,927 *unique* products purchased – constituting the "population" for our statistical analysis. Using an aggregation of the hospital's 218 expense accounts, we grouped these 2,927 products into 20 categories, 4 of which are excluded from our statistical analysis for the reasons given in Figure 8 and Appendix C. This categorization is ultimately arbitrary and is *not* a source of bias in our O-LCA results – provided that the categories are collectively exhaustive and mutually exclusive (which they are) and that the sampling within each category is truly random (which it is).

We assigned random numbers to all 2,927 unique products, thus allowing us to draw a statistically representative random sample of the products in each category. As the "supply-chains" dataset provides foreground data in both physical and economic values (*i.e.*, the physical quantities of products purchased, along with the cost per unit), we calculated the environmental intensity (*i.e.*, LCIA results per unit of expenditure) for each product in each category. We then calculated the *average* environmental intensity for each product category. This approach – despite the use of economic values – should *not* be confused with EEIO-LCA; the sole purpose of using monetary values in our approach is to express the environmental intensities of different products in common units (*e.g.*, kg CO_2 eq./$) as needed to calculate an *average* intensity value that serves as the population parameter to be estimated. Monetary values provide a convenient and universal denominator for this purpose. Using basic statistical inference (as detailed in Appendix C), we calculated the margin of error around the estimated average environmental intensity for each product category, and we multiplied the upper and lower bounds of the resulting confidence intervals by the total expenditures – for the corresponding product categories – to construct the confidence intervals for the total LCIA results for each product category.[8]

Notably, the "supply-chains" dataset does not cover purchases of pharmaceuticals, which are represented by relatively aggregated expense accounts in the hospital's financial statements.

[8] Technically, the *weighted* average, where the weights are the corresponding cost shares of the products, should be used. In our calculations, we use the *arithmetic* average (in kg CO_2 eq./$) for each of the 16 "supply-chain" product categories. This is done for two reasons: (1) to simplify the statistical analysis and (2) because the time period of the "supply-chains" dataset (*i.e.*, calendar year 2018) deviates slightly from the reference period of our O-LCA (*i.e.*, the hospital's 2019 fiscal year – April 1, 2018 through March 31, 2019). Nonetheless, to provide some indication of how the use of arithmetic – as opposed to weighted – averages might influence our results, we calculated the weighted averages for calendar year 2018, and we compare them to the arithmetic averages for the same time period (see Appendix C). For all 16 "supply-chain" product categories, the weighted averages are within the 99.7% confidence intervals (*i.e.*, the widest intervals) around the arithmetic averages. For the categories of "office supplies" (AC.02), "hardware supplies" (AC.09), "non-fuel heating supplies" (AC.11), "clinical nutrition" (AC.12), and "miscellaneous items C" (AC.19), the weighted averages are within the 68% confidence intervals (*i.e.*, the narrowest intervals) around the arithmetic averages. For the categories of "cleaning supplies" (AC.03), "medical products" (AC.04), "bedding supplies" (AC.05), "computer and electronic supplies" (AC.08), "non-drug pharmacy supplies" (AC.14), "laboratory supplies" (AC.16), and "sterilization supplies" (AC.18), the weighted averages are within the 95% confidence intervals.

Therefore, with the exception of anesthetic agents – for which we have foreground data on the physical quantities purchased during the reference period – the production of pharmaceuticals is represented using background data from the previously cited EEIO-LCA study of the Canadian healthcare sector (Eckelman et al., 2018). Thus, strictly speaking, our study can be considered a "hybrid" O-LCA – but one that is primarily bottom-up. This one exception – for pharmaceuticals – is necessary given the limitations of our foreground data, and, more importantly, background data (*i.e.*, there is presently a paucity of bottom-up LCA data on pharmaceuticals, though there are growing efforts to address this gap – see, *e.g.*, Belkhir & Elmeligi (2019), De Soete et al., (2017), Parvatker et al. (2019), and Siegert et al. (2019)).

For the LCIA phase, we use the *Tool for the Reduction and Assessment of Chemical and other environmental Impacts* (TRACI) method (version 2.1), which was developed under the purview of the U.S. EPA (Bare, 2011), is widely used in North American LCA studies, and is available for use in the OpenLCA (version 1.10.2) software used in our hospital O-LCA. We added global warming potential (GWP) characterization factors for anesthetic gases (*i.e.*, desflurane, isoflurane, and sevoflurane (Sherman et al., 2012)) as needed for our study.

In the O-LCA interpretation phase, where we assess the uncertainty attached to our O-LCA results, our statistical approach for the hospital's "supply-chains" created another practical challenge. To calculate the margins of error used to construct confidence intervals around the O-LCA results for each of the 16 "supply-chain" categories, it is necessary to calculate the standard deviation of the environmental intensities of the sampled products in each category. Calculating the standard deviation for a product category requires the environmental intensity of *each product* in the category (along with the category average) to be calculated first. As to our knowledge OpenLCA does not have the functionality to calculate the standard deviations internally (and we suspect the same limitation also applies to other LCA software programs, given the novelty of this calculation approach in LCA), we exported our O-LCA results from OpenLCA into a spreadsheet (see Appendix C) to calculate the standard deviations (and ultimately the margins of error). With a total of 16 “supply-chain” product categories, each represented by its own worksheet, calculating the margins of error for all 10 TRACI impact categories would require 160 worksheets. For practical reasons, therefore, we report confidence intervals for GWP only, which nonetheless provide an indication of the uncertainty attributable to our sampling approach.

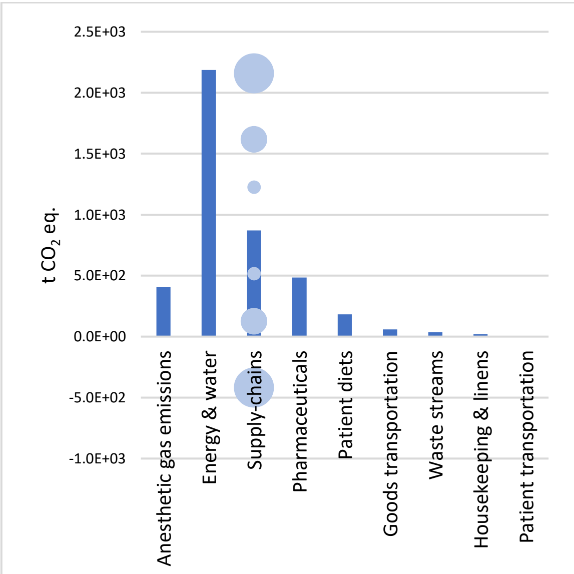
4.4 Results and discussion

Previous work on environmental footprinting in healthcare often provides results for a limited set of environmental impact categories (Drew et al., 2021), with particular fixation on GWP (see, *e.g.*, the top-down studies by Malik et al. (2018), Nansai et al. (2020), Tennison et al. (2021), and Wu (2019)). In contrast, we report O-LCA results for the full suite of TRACI impact categories (*i.e.*, GWP, acidification potential, eutrophication potential, smog formation potential, ozone depletion potential, human toxicity – respiratory effects, human toxicity – carcinogenic effects, human toxicity – non-carcinogenic effects, ecotoxicity, and fossil fuel depletion). However, it is important to remember that our O-LCA results are only representative of a single year of operation, of a single hospital, prior to the COVID-19 pandemic. Nonetheless, our study is, to the best of our knowledge, the first application of O-LCA in the healthcare sector, and to date the most comprehensive bottom-up environmental footprinting study of a healthcare facility.

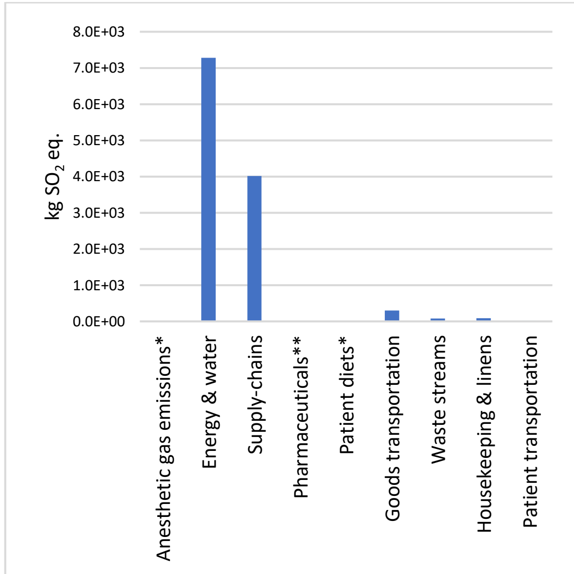
The novelty of our hospital O-LCA precludes direct comparisons with previous work, like the top-down studies cited in the introduction to this article (Eckelman et al., 2018, 2020; Eckelman & Sherman, 2016; Malik et al., 2018; Tennison et al., 2021; Wu, 2019). Perhaps the closest comparison – which still needs to be done cautiously, given differences in context, scope, methods, and data – is with the "hybrid" LCA, of an inpatient unit and an ICU, by Prasad et al. (2022). With that in mind, and drawing from our O-LCA results, we make four broad observations about the scale, complexity, and variability of the environmental footprint of healthcare facilities.

A first observation is that, in terms of scale, our results indicate that the total GWP of the hospital's operations – in its 2019 fiscal year – is in the range of 3,500 to 5,000 t CO_2 eq., with 95% confidence. For comparison, Prasad et al. (2022) found that the inpatient unit and ICU at their studied hospital generated about 656 t CO_2 eq. and 349 t CO_2 eq., respectively, over the one-year period covered by the study. Therefore, the total GHG emissions from the inpatient unit (with 49 beds) and ICU (with 12 beds) combined are approximately 1,000 t CO_2 eq. This number is notably below the estimated total for the hospital in our study (with a total of about 40 beds). There are many factors that could explain this apparent discrepancy (*e.g.*, differences in the hospitals studied, the study scope, data sources, and other methodological aspects), hence the results are not directly comparable. Nonetheless, both estimates – of the operations of a hospital (or part thereof) over a one-year period – are roughly on the same order of magnitude.

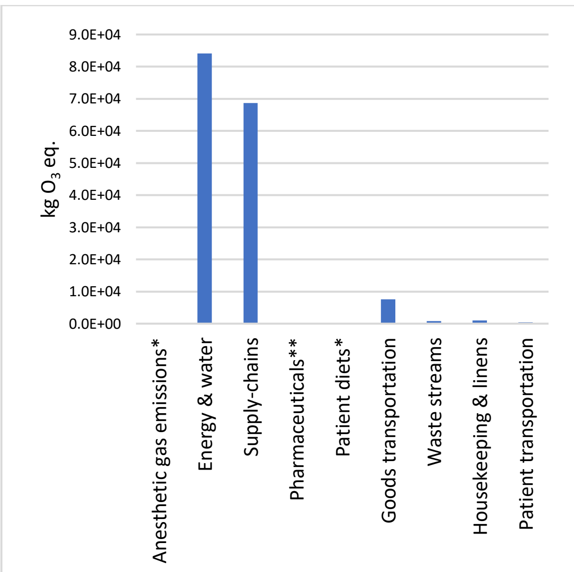
A second observation – more consistent with the findings of Prasad et al. (2022), as can be seen in their Figure 2 – is that energy and water use (including sewerage) account for a major "hotspot" in the hospital's environmental footprint; this finding is consistent across all impact categories (see our Figure 9).



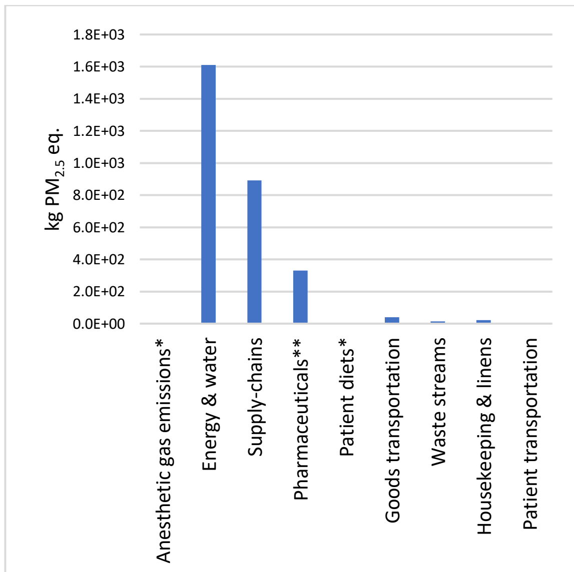
A. Global Warming Potential (GWP)



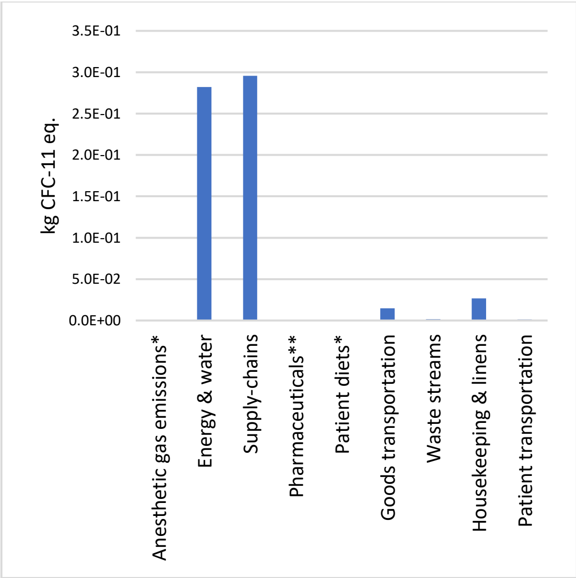
B. Acidification Potential



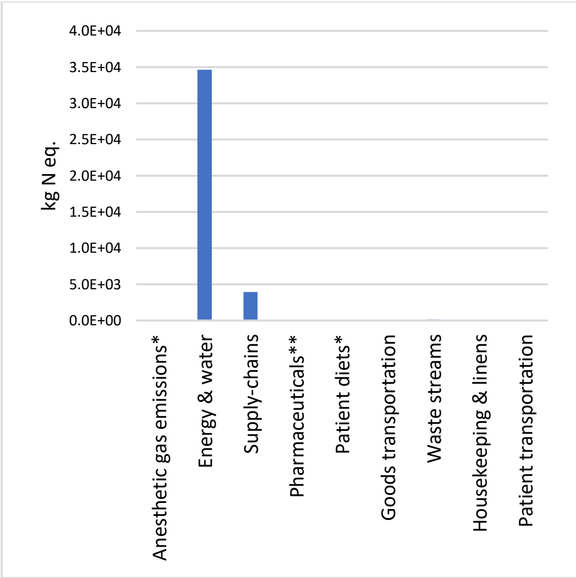
C. Smog Formation Potential



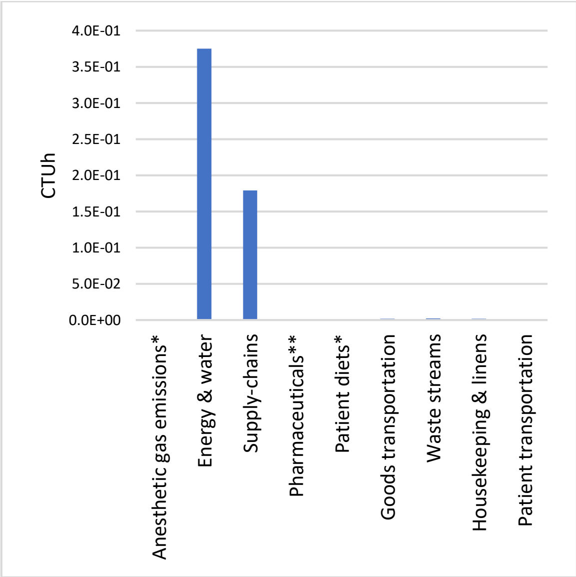
D. Respiratory Effects



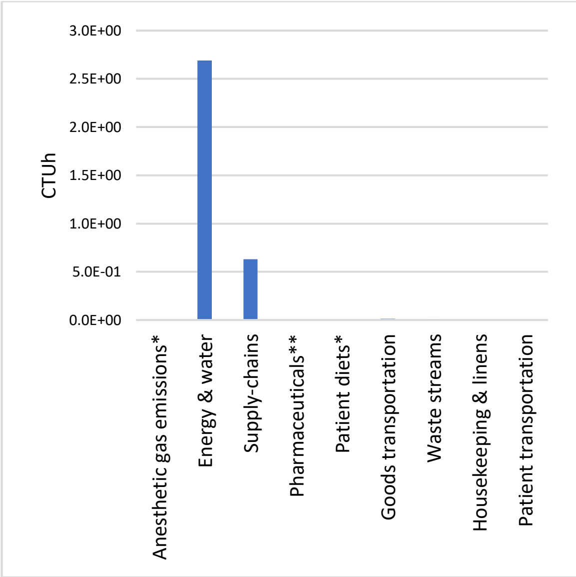
E. Ozone Depletion Potential



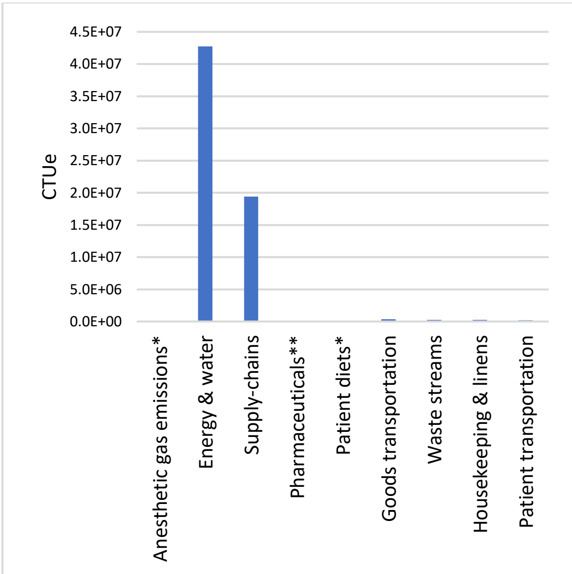
F. Eutrophication Potential



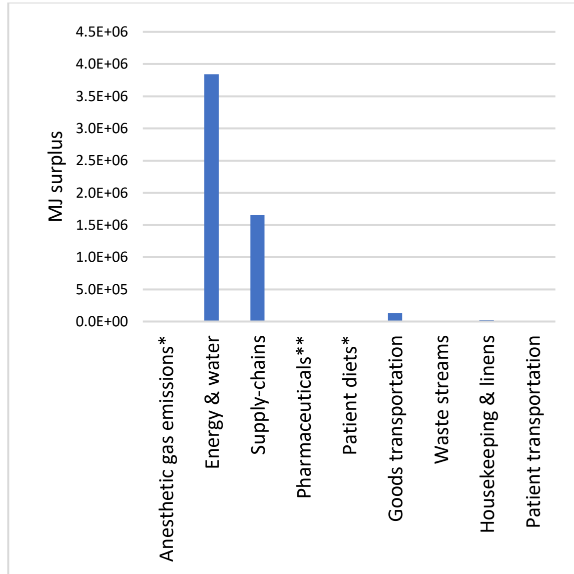
G. Human Toxicity, Carcinogenic



H. Human Toxicity, Non-Carcinogenic



I. Ecotoxicity



J. Fossil Fuel Depletion

*The underlying data supports O-LCA results for only GWP.
**The underlying data supports O-LCA results for only GWP and Respiratory Effects.

Figure 9: "Environmental footprint" of the hospital's operations in its 2019 fiscal year.[9]

The environmental footprint of energy and water use can also vary significantly depending on the usage rates (which vary over time) and the upstream processes (*e.g.*, in the electricity supply mix, which also varies over time and by location). Therefore, recognizing that our studied hospital is in an area with a relatively low-carbon electrical grid, we consider how our results could change in the hypothetical scenario of an identical hospital in Alberta – a neighbouring province with a higher-carbon grid (Canada Energy Regulator, 2021b). This scenario, as detailed

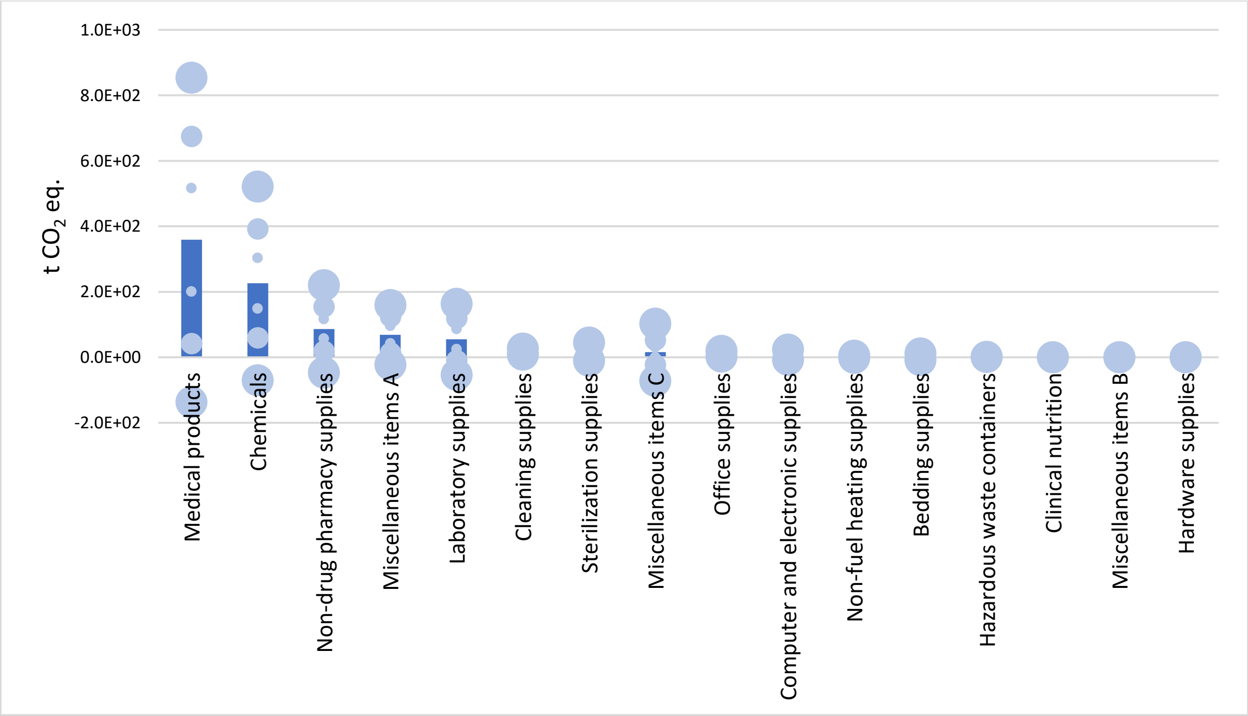
[9] The coloured dots around the GWP contribution of "supply-chains" represent confidence intervals at different confidence levels, with heavier dots indicating higher confidence. The smallest dots represent the 68% confidence interval (*i.e.*, the tightest interval), the medium-sized dots represent the 95% confidence interval (*i.e.*, a wider interval), and the heaviest dots represent the 99.7% confidence interval (*i.e.*, the widest interval). For practical reasons explained in the text, we report confidence intervals for GWP only. It is also important to note that these confidence intervals account for only the uncertainty attributable to random sampling; they do not account for the uncertainty attributable to limitations of the quality of the underlying data used to represent the products in the sample.

in Appendix C, more than doubles the GWP contribution of the hospital's energy and water use, and therefore increases the hospital's total GWP to 6,600-8,000 t CO_2 eq., with 95% confidence.

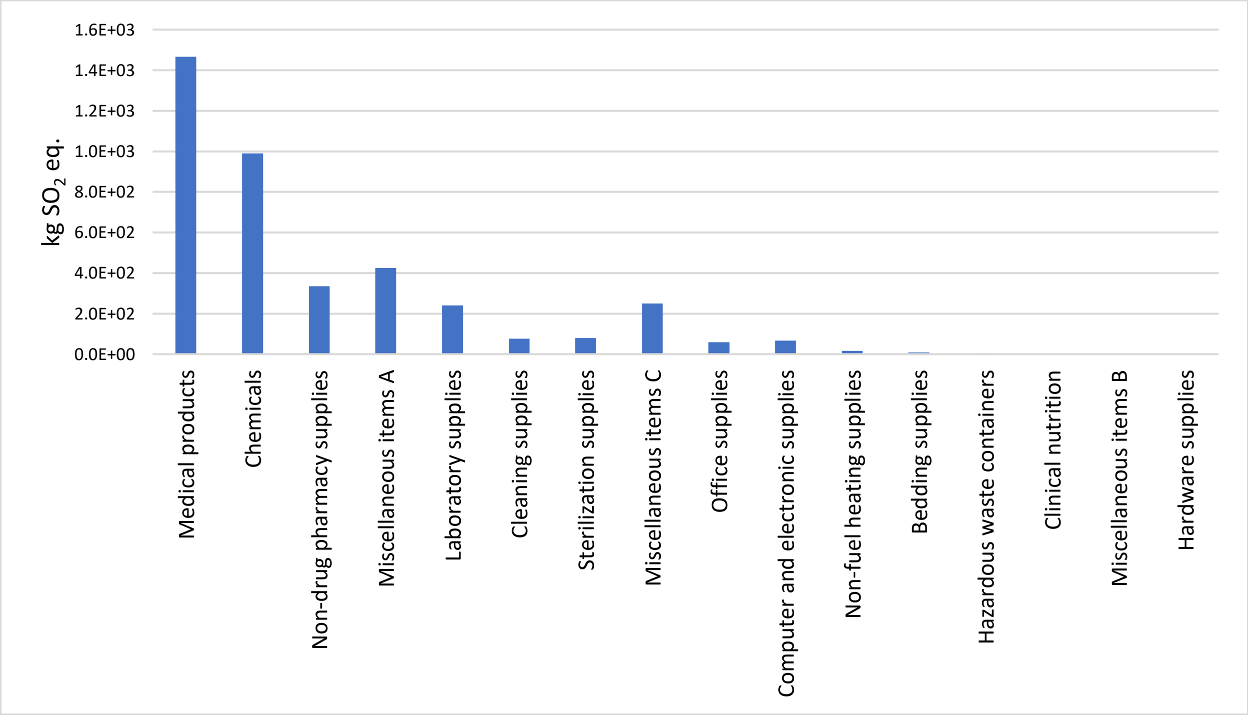
A third observation – also consistent with the findings of Prasad et al. (2022) – is that transportation (of patients and of goods) and waste management – despite being popular and visible focus areas in the domain of "healthcare sustainability" (as seen *e.g.* in our scoping review of the academic literature (Cimprich et al., 2019), and in practical initiatives like those organized through the Canadian Coalition for Green Health Care (see link in reference list) and the international nongovernmental organization named Health Care Without Harm (see link in reference list)) – contribute relatively little to the hospital's total environmental footprint. We note that, along with the category of "goods transportation" (modelled using aggregated expenses documented in the hospital's financial statements), we also embedded transportation (from first-tier suppliers, modelled using confidential supplier information) in our LCI data for the products in the "supply-chains" dataset. This is done to give an indication of the relative contribution of transportation versus production of the products (as can be seen in Appendix C). Though it may result in some double-counting, this would not significantly change our overall O-LCA results, as the "goods transportation" category makes only a minor contribution to all impact categories (see Figure 9 and Appendix C). On the other hand, we also note that the footprint of goods transportation could increase significantly if air freight is used (Horvath, 2006); our default assumption is that all shipping is done via trucks. Further, as previously noted, we omit employee travel and commuting (which was included in the Prasad et al. (2022) study, where it was found to make a relatively minor contribution to the total GHG emissions), along with patient and visitor travel using vehicles not owned or controlled by the hospital or its contracted patient transportation (*i.e.*, ambulance) service providers.

A fourth observation – again consistent with the Prasad et al. (2022) study – is that a complex set of hotspots is attributable to less visible aspects of the hospital's operations, besides the obvious considerations of energy and water use, transportation, and waste management. Most notably, as can be seen in Figure 9, the hospital's "supply-chains" – comprising the purchases of 2,927 unique products – contribute a major hotspot with respect to most impact categories. In fact, our statistical analysis indicates that, at the 99.7% confidence level, the contribution of "supply-chains" – at least with respect to GWP – could be on par with energy and water use (Figure 9A).

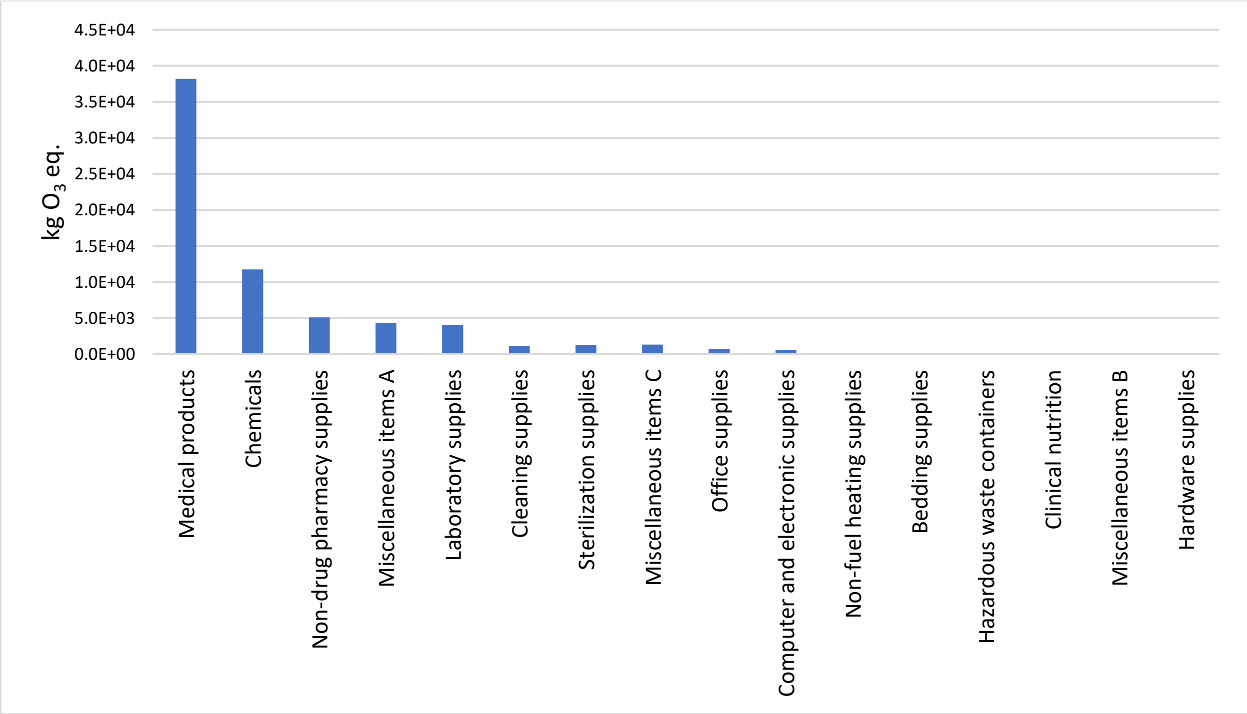
At this point, we highlight one of the most important contributions of our hospital O-LCA. Along with providing environmental footprint results beyond those associated with GHG emissions (as is good practice in LCA, though commonly omitted in the literature on healthcare sustainability), our mostly bottom-up approach does what top-down approaches cannot – it provides detailed data and results for *specific* goods and services (beyond energy and water utilities, waste management, and transportation) used in the hospital. As can be seen in Figure 10, the contributions of the 16 "supply-chain" categories vary significantly between each other and between impact categories. In particular, the broad and diverse category of "medical products," itself comprising 1,683 unique products (*i.e.,* more than half of all unique products in the hospital's "supply-chains"), is among the largest contributors to most impact categories. The contribution of things like "office supplies," on the other hand, is comparatively small.



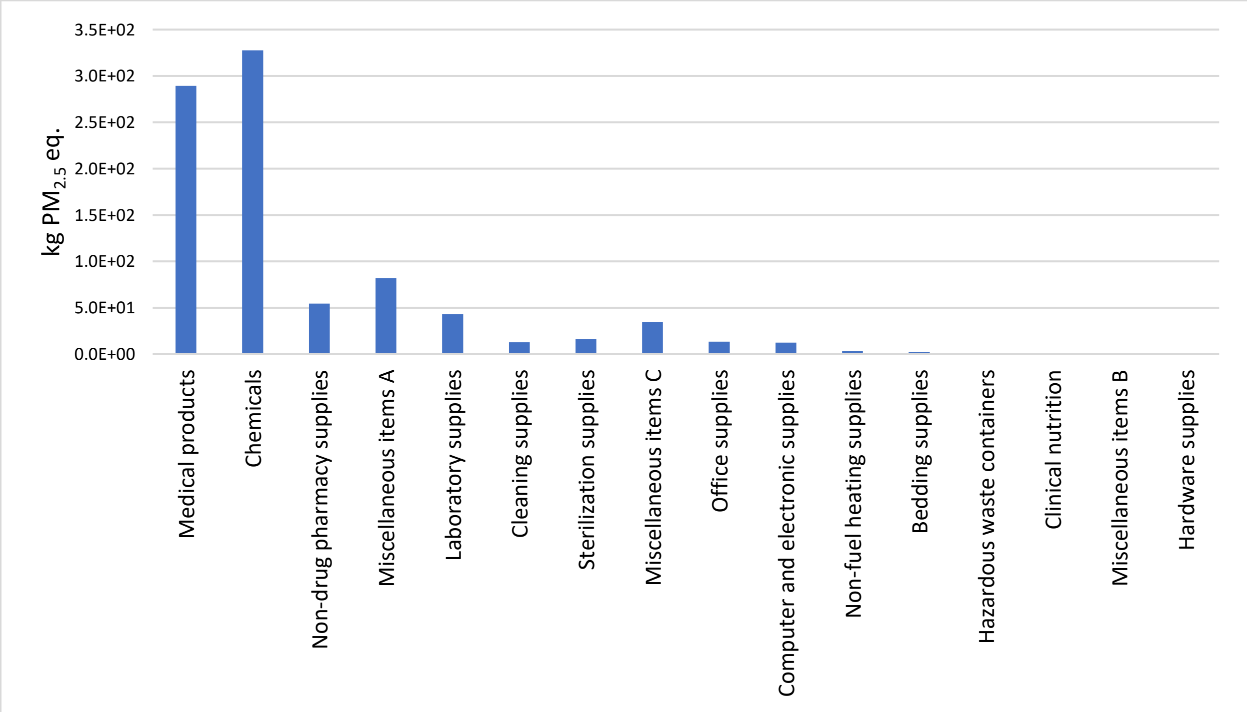
A. Global Warming Potential (GWP)



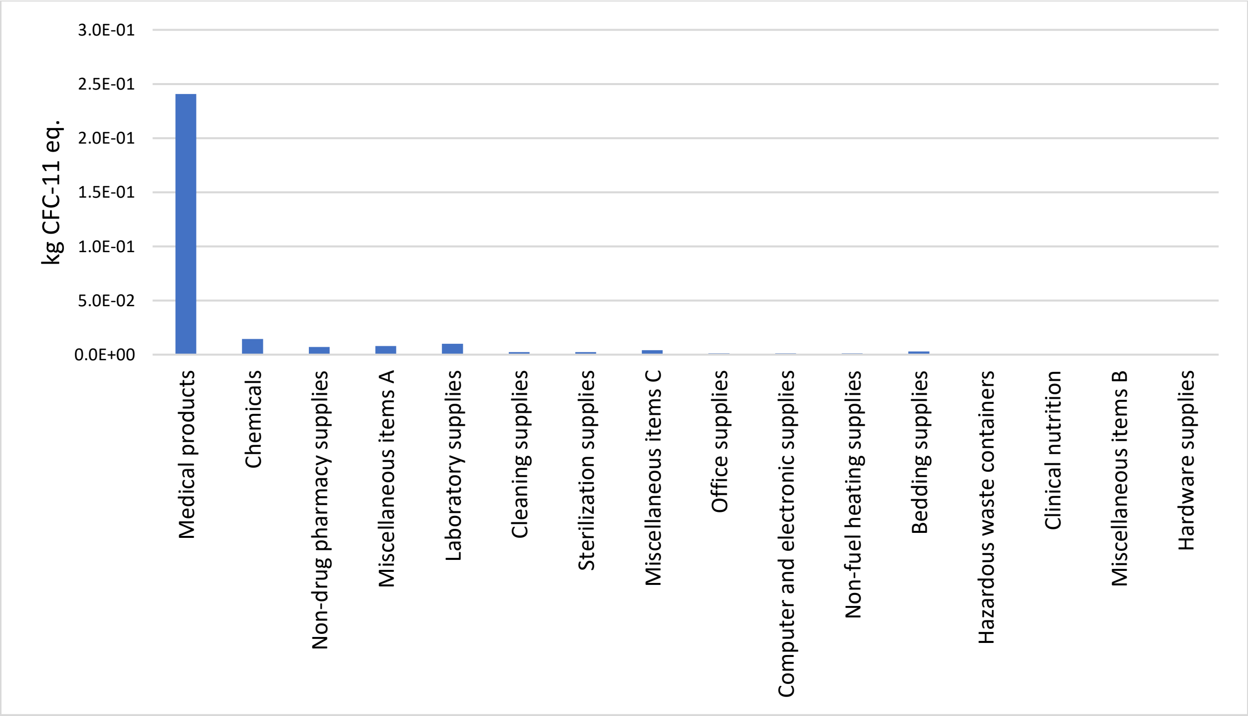
B. Acidification Potential



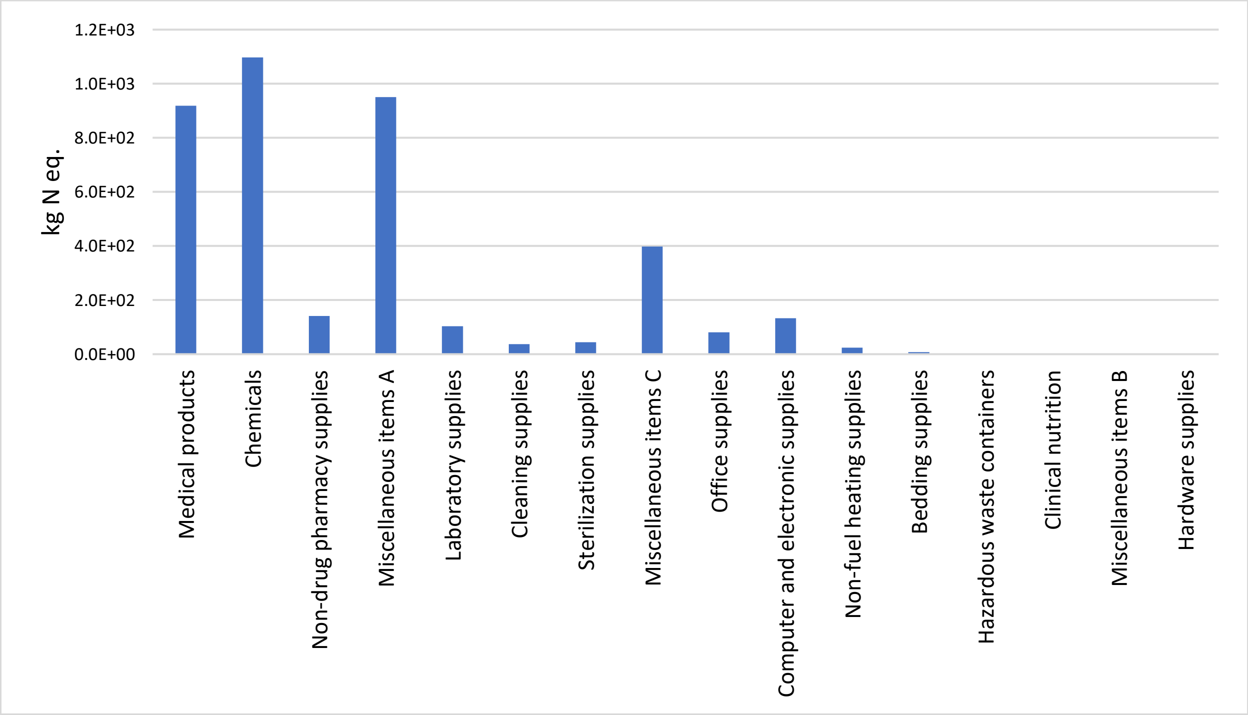
C. Smog Formation Potential



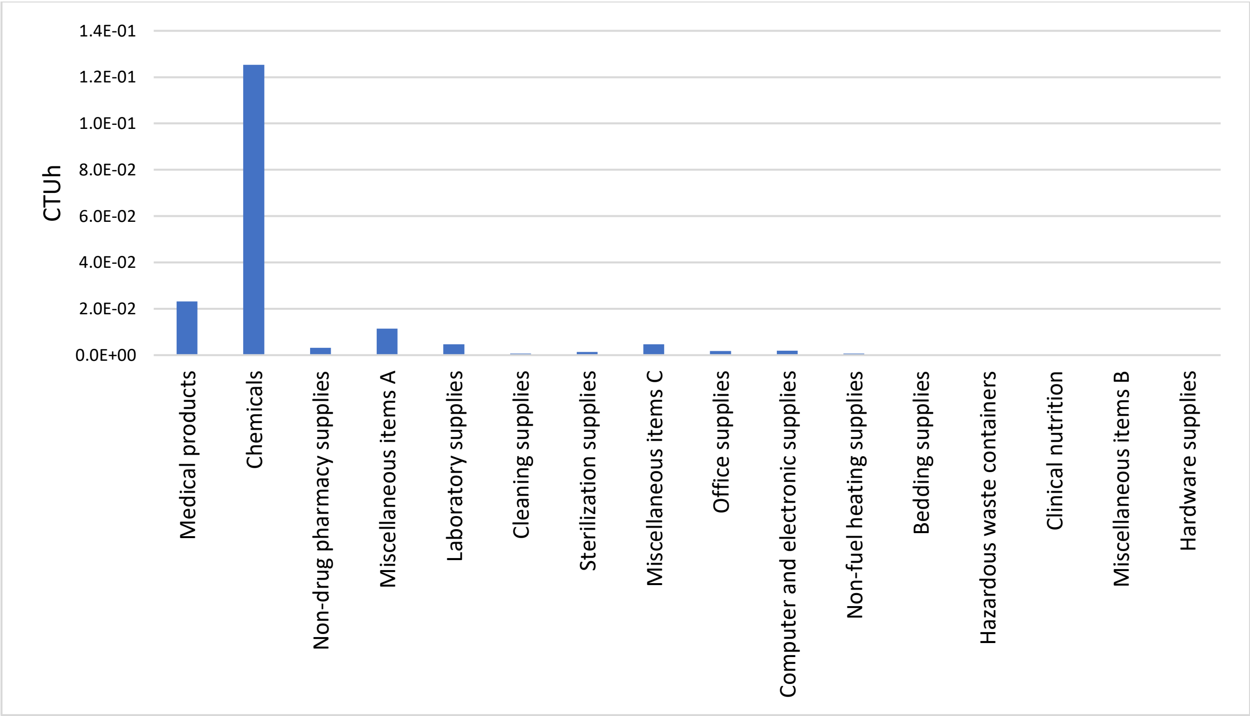
D. Respiratory Effects



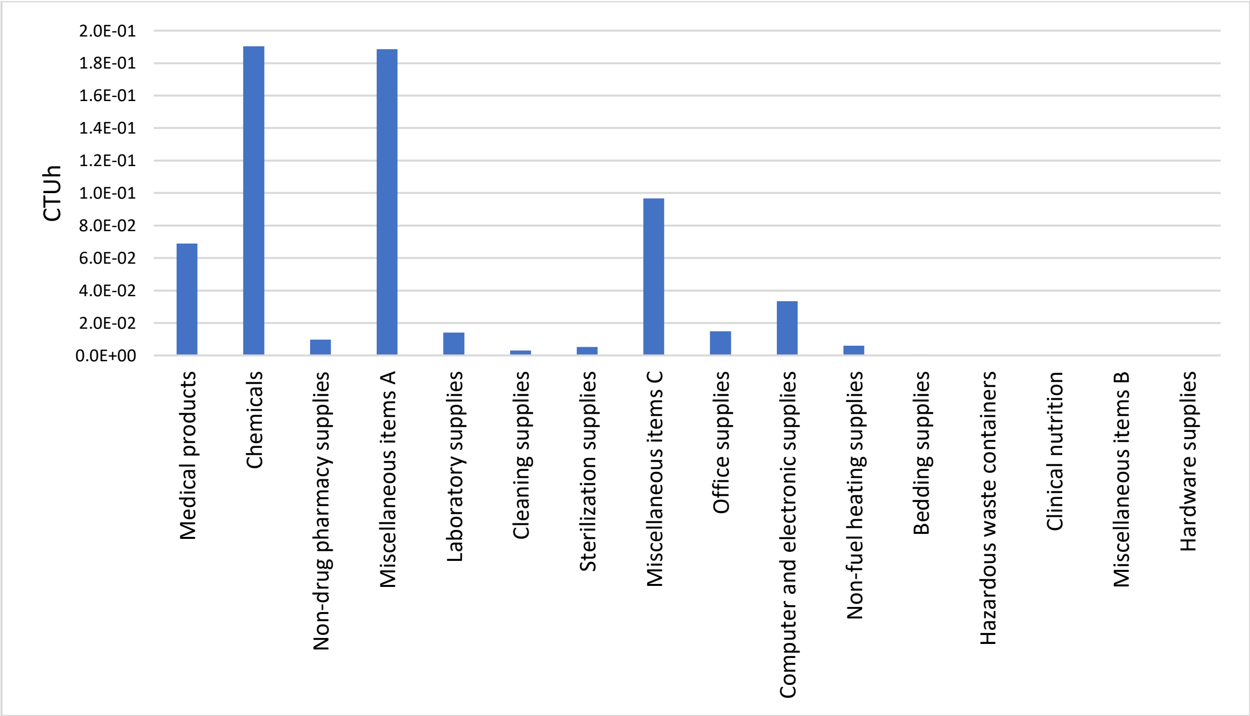
E. Ozone Depletion Potential



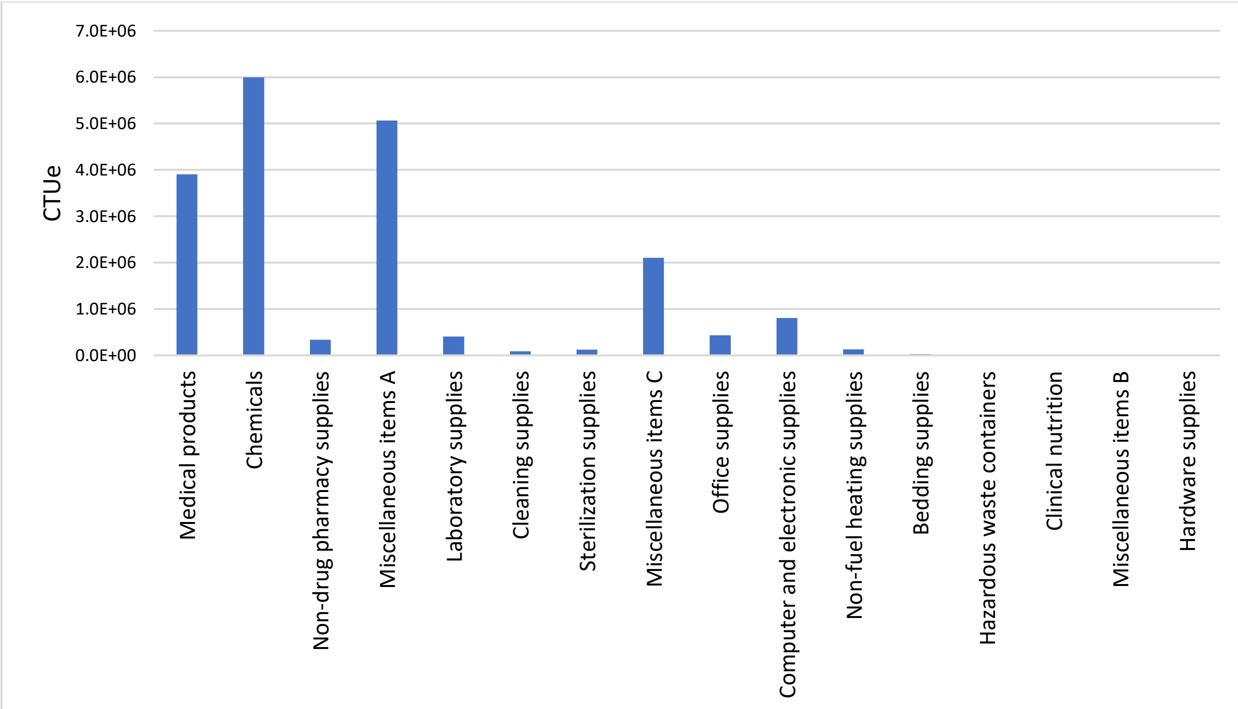
F. Eutrophication Potential



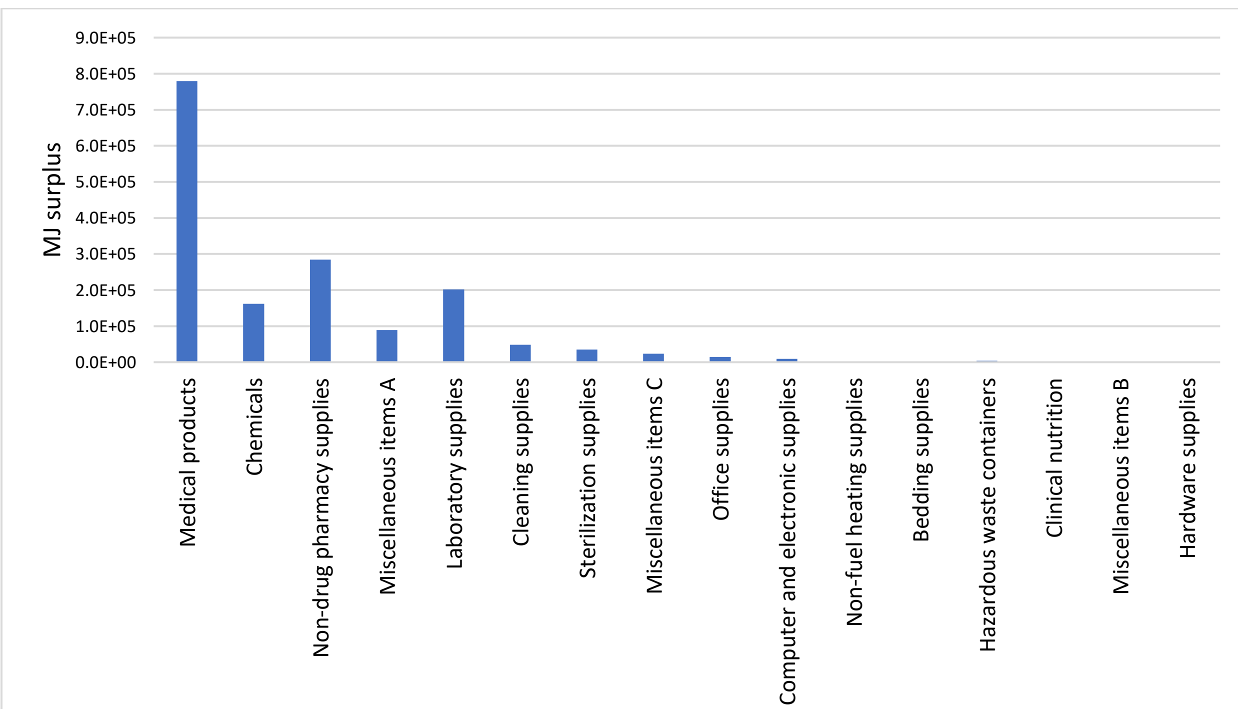
G. Human Toxicity, Carcinogenic



H. Human Toxicity, Non-Carcinogenic



I. Ecotoxicity



J. Fossil Fuel Depletion

Figure 10: "Environmental footprint" of products purchased through the hospital's "supply-chains."[10]

Further granularity is provided in Appendix C, which contains a detailed breakdown of the O-LCA results for each of the approximately 200 goods and services modelled in our hospital O-LCA. The contributions of the products purchased through the hospital's "supply-chains" vary by several orders of magnitude. Among these products, Table 2 highlights some of the largest and smallest contributors with respect to the hospital's total environmental footprint (*i.e.*, based on the total quantities of these products purchased in the 2018 calendar year). As our O-LCA results – including those for the hospital's "supply-chains" – are generated almost entirely through a bottom-up approach, it is possible to probe even deeper into the data and results associated with each product (as documented in Appendix C), although for the "supply-chains" only a relatively small *sample* of the 2,927 unique products were modelled in our study.

[10] The coloured dots around the GWP contributions of the "supply-chain" product categories represent confidence intervals at different confidence levels, with heavier dots indicating higher confidence. The smallest dots represent the 68% confidence interval (*i.e.*, the tightest interval), the medium-sized dots represent the 95% confidence interval (*i.e.*, a wider interval), and the heaviest dots represent the 99.7% confidence interval (*i.e.*, the widest interval). For practical reasons explained in the text, we report confidence intervals for GWP only. It is also important to note that these confidence intervals account for only the uncertainty attributable to random sampling; they do not account for the uncertainty attributable to limitations of the quality of the underlying data used to represent the products in the sample.

Table 2: Examples of total "environmental footprint" contributions of products in the hospital's "supply-chains"

Product description	UNSPSC® commodity code	Quantity purchased in calendar year 2018	Contribution as a percentage of the hospital's total "environmental footprint" in its 2019 fiscal year									
			GWP	AP	Smog	Resp	ODP	EP	Carc	Non-Carc	Ecotox	FD
Medicine cups	42192603-01	623 packages of 5,000 each	0.25%	0.33%	0.30%	0.23%	0.08%	0.05%	0.07%	0.03%	0.06%	0.66%
Medical tubing	42000000-02	1,163 sets weighing 2.6 kg each	0.19%	0.23%	0.27%	0.12%	0.03%	0.02%	0.09%	0.04%	0.06%	0.38%
Surgical custom packs	42295414-01	869 packs	0.18%	0.00%[b]	0.00%[b]	0.00%[b]	0.03%	0.07%	0.00%[b]	0.00%[b]	0.00%[b]	0.00%[b]
Sharps containers	42142531-01	1,934 containers in total (including several varieties)	0.08%	0.10%	0.09%	0.07%	0.03%	0.01%	0.02%	0.01%	0.02%	0.21%
Oxygen cylinders	42271701-01	22 cylinders in total (including two varieties)	0.06%	0.10%	0.09%	0.14%	0.03%	0.03%	0.28%	0.07%	0.12%	0.03%
Medical surface disinfectants	42281604-02	86 cases of four 5 L containers	0.04%	0.06%	0.05%	0.05%	0.05%	0.01%	0.04%	0.01%	0.03%	0.07%
Urine collection containers	41104112	1,086 containers	0.04%	0.06%	0.05%	0.04%	0.01%	0.01%	0.01%	0.01%	0.01%	0.11%
Blood pressure cuffs	42181605	17 disposable and 4 reusable cuffs	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]
Medical or surgical gloves	42132203-01 & 42132205-01	2,303 gloves	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]
Intraocular lenses	42295505-01	106 lenses	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]	0.00%[a]

GWP = Global Warming Potential, AP = Acidification Potential, Smog = Smog Formation Potential, Resp = Respiratory Effects, ODP = Ozone Depletion Potential, EP = Eutrophication Potential, Carc = Human Toxicity (Carcinogenic), Non-Carc = Human Toxicity (Non-Carcinogenic), Ecotox = Ecotoxicity, FD = Fossil Fuel Depletion

[a]Contribution percentage < 0.01%

[b]Missing LCI data and/or characterization factor(s)

Along with other limitations of our hospital O-LCA, as previously noted (*e.g.*, that our results are only representative of a single year of operations of a single hospital, that we used a "top-down" approach for pharmaceuticals, and that we excluded employee travel and commuting), it is also important to acknowledge the limitations of our "matching" between foreground and background data. Our background data are not perfectly representative of the goods and services purchased by the hospital; the potential *mis*matching between foreground and background data is particularly noteworthy for the products purchased through the hospital's "supply-chains," especially given the deficiencies in the quantity *and* quality of healthcare-related LCAs, as evidenced by the systematic review by Drew et al. (2021) cited in the introduction to this article. It is worth noting, however, that the granularity of our bottom-up approach, using foreground data on *specific* goods and services purchased by the hospital, reduces this potential source of error compared to a top-down approach, using foreground data on aggregated expenditure categories. The more granular the foreground data, the better the ability to evaluate the representativeness of the corresponding background data (*e.g.*, it is easier to model "medicine cups," "sharps containers," or "blood pressure cuffs" than to model "medical products"). On the other hand, it is also worth noting that the statistical confidence intervals around our GWP results account for only the uncertainty attributable to random sampling; they do not account for the uncertainty attributable to the quality of the underlying data – especially background data – used to derive our O-LCA results.

Again, we need to emphasize that the limitations of existing healthcare-related LCAs ultimately translate into limitations of the background data for our hospital O-LCA. Further, it is important to note that we have not conducted a systematic and comprehensive evaluation of data quality – for either foreground or background data – in our hospital O-LCA (*e.g.*, with respect to geographical and technological representativeness, and the internal consistency of methodological aspects like allocation methods). Such an evaluation requires extensive knowledge of the products the background data are intended to represent (*i.e.*, to evaluate the representativeness *of what*?), along with transparent documentation of the methodology of each background data source – both of which are often lacking. For example, despite the high quality of our foreground data on the hospital's "supply-chains" (which indicates the exact quantities purchased, in physical and monetary units, of each specific product from each specific supplier), it is often unclear exactly where each product is produced (*e.g.*, where the supplier has a broad portfolio of products produced in multiple facilities), what materials it is made from (necessitating loosely educated assumptions based on the personal knowledge and experience of the authors, combined with product descriptions and imagery where available), and with what manufacturing technologies. The problem is compounded by the present reality that, as indicated by our review and that of Drew et al. (2021), healthcare-related LCAs are in their infancy, and manufacturers of healthcare-related products rarely provide basic information like product mass and material composition, much less something akin to environmental product declarations (EPDs) based on rigorous and transparent LCA studies. Moreover, even with the aid of our statistical sampling approach, there are still hundreds of products for which a highly context-dependent data quality assessment would need to be conducted. Even then, it may be challenging to quantify the overall uncertainty in the O-LCA results, considering *both* sources of uncertainty – *i.e.*, the uncertainty attributable to random sampling and the uncertainty attributable to data quality limitations.

Nonetheless, compared to previous efforts to quantify the environmental footprint of healthcare, the new level of comprehensiveness and granularity in our hospital O-LCA – particularly with respect to healthcare "supply-chains" – helps provide more actionable information to support decision-making in healthcare facilities (*e.g.*, in energy and facilities management, procurement, and clinical practice).

4.5 Conclusion

Our hospital O-LCA makes three contributions to industrial ecology and "healthcare sustainability." First, it constitutes, to the best of our knowledge, the most comprehensive bottom-up assessment of the environmental footprint of a whole hospital conducted to date. Second, it does so by using a new methodological approach – O-LCA (with a novel application of basic statistical sampling and inference techniques) – for environmental footprinting that shows promise for overcoming, at least to some extent, the present trade-off between the comprehensiveness of top-down approaches and the granularity of bottom-up approaches. Finally, in the process of conducting our hospital O-LCA, we have compiled new LCI data for approximately 200 goods and services used in healthcare. Our work thus aligns strongly with the criteria given by Lifset (2013) for "raising the bar" in LCA case studies.

Future work could apply a similar O-LCA methodology to other hospitals, and to other types of healthcare facilities (and other organizations beyond the healthcare sector), while assessing their environmental footprint over multiple years – to support benchmarking across facilities and across time. Another, related direction could be to scale the O-LCA methodology to assess the environmental footprint of larger healthcare systems – *e.g.*, the entire regional *health authority* (*i.e.*, a larger organization – one of four in British Columbia's lower mainland – that administers publicly funded healthcare services to a population of over 1 million people). This approach is likely more feasible than it may seem, for three reasons: (1) in our case, many of the foreground data sources, including the "supply-chains" dataset, are universal across all facilities within the health authority, (2) there is likely to be considerable overlap in the *unique* products purchased by each facility, and (3) given the fundamental principles of statistical inference, the margin of error – around the estimated average environmental intensity of the products purchased through the health authority's "supply-chains" (see the equation provided in Appendix C) – depends mainly on the *sample* size rather than the *population* size (where the term "population" in this context refers to the full set of all unique products purchased). Yet another direction could be to "zoom in" on the hotspots identified in our O-LCA – particularly to more closely examine the broad and diverse category of "medical products" (which comprises more than half of the 2,927 unique products in the hospital's "supply-chains"). Further research along these lines is essential to improve the coverage and quality of LCA data for the myriad of products used in the provision of healthcare services. Finally, recognizing that the statistical approach we used to model the hospital's "supply-chains" (*i.e.*, confidence intervals for means, in statistical terms) is quite rudimentary, we see opportunities to explore the potential for more advanced statistical methods (*e.g.*, regressions or other techniques to predict or impute the environmental intensities of products beyond those included in the sample) to refine our approach.

Supplementary content

See Appendix C.

Data Availability Statement

Aside from the data contained in Appendix C, other "foreground data" used in our study (*e.g.*, from the hospital's purchase records and financial statements) are confidential. "Background data," for approximately 200 goods and services used in healthcare, include a mix of new data generated in our study, and existing data from a third party (*i.e.*, from the Ecoinvent 3.4 database); the former are available in Appendix C, and the latter are available from Ecoinvent.

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Chapter 5

5. Conclusion

Altogether, this thesis makes several conceptual, methodological, and practical contributions in relation to *sustainability management*. Conceptual contributions include an industrial ecology framework for “healthcare sustainability” (as depicted in Chapter 2, Figure 4) and a conceptualization of organizational life cycle assessment (O-LCA) in relation to other widely used analytical approaches for environmental sustainability (as depicted in Chapter 3, Figure 6). Methodological contributions arise from the novel use of basic statistical sampling and inference techniques in two distinct applications: (1) a scoping review of the literature on healthcare sustainability (see Chapter 2), and (2) a constructive critique of organizational LCA (see Chapter 3) with a hospital case study (see Chapter 4). These conceptual and methodological contributions are operationalized through practical contributions aimed at supporting *sustainability management* – including the hospital O-LCA (as to date the most comprehensive and detailed assessment of the environmental footprint of a healthcare facility), and the new LCA data compiled for approximately 200 goods and services used in healthcare. Though my focus has been on sustainability management in healthcare – a critical and complex service sector that has only recently begun to grapple with sustainability – the contributions in this thesis can also support sustainability management in other sectors. In the following paragraphs, I briefly comment on each of these contributions – summarizing and extending what has already been said in previous chapters – and then highlight opportunities for future work building upon this thesis.

Regarding the conceptual contributions from this thesis, the industrial ecology framework for healthcare sustainability (as depicted in Chapter 2, Figure 4) – which combines ideas from myself and my co-authors – integrates three key features: (1) the notion of "foreground" and "background" systems, (2) the notion of "medical" and "supporting" activities, and (3) the notion of "inside-out" and "outside-in" impacts. As I will elaborate shortly, these ideas are largely, albeit not fully, operationalized in the hospital O-LCA presented in Chapter 4. In Chapter 3 (Figure 6), my co-author and I conceptualized O-LCA in relation to other widely used analytical approaches for environmental sustainability (particularly product LCA, EEIO-LCA, and organizational carbon footprinting) in terms of comprehensiveness (*i.e.*, with respect to system boundaries and the types of environmental impacts assessed) and granularity (*i.e.*, the level of detail to which the environmental impacts can be disaggregated to highlight "hotspots" in the system).

Regarding methodological contributions, there is nothing new about the basic statistical sampling and inference techniques (*i.e.*, random sampling and confidence intervals) used in the healthcare sustainability scoping review (Chapter 2) and the methodological proposals for O-LCA (Chapter 3) tested and demonstrated through the hospital case study (Chapter 4). The novelty lies in the implementations of these methods. Though the context is different, the fundamental methodological problem – in both the scoping review and in the O-LCA – is the same: there is a *population* with an unknown *population parameter* to be estimated based on a *sample* from the population. In the context of the scoping review, the "population" comprises the 1,748 articles collected through systematic literature searches, and the population parameter to be estimated is the percentage of those articles meeting certain criteria (*i.e.*, fitting within certain categories describing the contents of the articles). In the hospital O-LCA, the "population" comprises the

2,927 unique products purchased through the hospital's supply-chains (which were grouped into 20 "sub-populations"), and the (sub)population parameter to be estimated is the average environmental intensity of the products in each sub-population. In my view, these unconventional applications of basic statistical sampling and inference techniques reinforce the general principle that methodological choices should be guided by the *specific* aims and objectives of the research in question (see, *e.g.*, Creswell (2014)). I am not suggesting that every scoping review should use a statistical approach like that applied in Chapter 2, but for the purpose of that *particular* scoping review the statistical approach made sense. Likewise, in an O-LCA where there are thousands of products in the system boundary, and it is not feasible to conduct LCAs for all of them, some form of sampling is unavoidable (here, I note that the "representative products" approach used in the O-LCA of Natura & Co. (de Camargo et al., 2019) is also a form of sampling, whether or not the authors of that study explicitly use terms like "population" and "sample"). Obviously, it is desirable to make the sample as representative as possible, and to be able to measure and evaluate – ideally in a quantitative way – just how "representative" it is. That is precisely what statistical inference is all about, and precisely where the "representative products" approach exemplified by the Natura & Co. study goes wrong.

Regarding practical contributions, the healthcare sustainability scoping review (Chapter 2), admittedly, is of little help to those "on the ground" in healthcare – particularly clinicians, managers, and administrators. It may also become outdated with respect to the rapidly evolving area of healthcare sustainability. Nonetheless, it highlights the need for a more systematic and evidence-based approach in a critical and complex service sector that, until recently, has been largely overlooked with respect to sustainability management. It is also grounded in an industrial ecology framework for healthcare sustainability, which is to a large extent operationalized through the hospital O-LCA presented in Chapter 4. With respect to the healthcare sustainability framework, the hospital is a “foreground” system in which “medical” activities (*e.g.*, medical imaging and surgical procedures) and “supporting” activities (*e.g.*, hospital housekeeping, maintenance, and food services) take place. Through its medical and supporting activities, the hospital has a multitude of inputs and outputs with respect to “background” systems (*e.g.*, for the production of the 2,927 unique products purchased through the hospital’s supply-chains, and for the treatment of the hospital’s waste streams). The foreground and background systems have “inside-out” impacts with respect to environmental sustainability (*i.e.*, in terms of GWP, acidification potential, eutrophication potential, and so on). Other sustainability dimensions (*i.e.*, social and economic), along with “outside-in” impacts, are beyond the scope of the study. Nonetheless, recognizing that the O-LCA results are only representative of a single year of operations of a single hospital, the hospital O-LCA bridges the gap between “top-down” studies of healthcare on a national-sector-level (which are too aggregated to inform “on the ground” decision-making) and “bottom-up” studies of specific healthcare-related products and procedures (which are very limited in scope). The mostly bottom-up approach, on a much larger scale than a typical bottom-up study, of the hospital O-LCA yields results with a level of comprehensiveness and detail not seen in previous work – thus providing more actionable information to support decision-making. To that end, in Appendix D the O-LCA results are re-expressed in relation to the broad categories of “Scope 1, Scope 2, and Scope 3” per the *GHG Protocol Corporate Standard* (WRI & WBCSD, 2004), which is widely used in practical efforts towards healthcare sustainability (as seen in, *e.g.*, the work of the U.K. National Health Service (NHS; see link in reference list) and the international nongovernmental organization named Health Care Without Harm (see link in reference list)). Moreover, in the process of conducting the O-LCA, new LCA data were compiled for approximately 200 goods and services used in healthcare – at least some of which may also be used in other sectors.

Building upon these contributions, Chapter 4 of this thesis also highlighted opportunities for future work, including:

- O-LCAs of other hospitals, other types of healthcare facilities, and organizations in other sectors;
- O-LCAs covering multiple years of operation;
- O-LCAs of larger organizations, such as the entire regional "health authority" to which the hospital studied in Chapter 4 belongs;
- "Zooming in" on "hotspots" identified in the hospital O-LCA, such as the broad and diverse category of "medical products" purchased through the hospital's supply-chains;
- Improving the coverage and quality of LCA data for healthcare-related products; and
- Further refining the O-LCA methodology, particularly by using more advanced statistical techniques to better predict or impute the environmental impacts of products beyond those included in the representative sample.

Ultimately, all these efforts are aimed at building a stronger evidence base to support decision-making – as the essence of *sustainability management* – in healthcare and in all other sectors.

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Appendices

Appendix A

Supplementary content for Chapter 2

The following supplementary files are available free of charge on the journal's website:

https://doi.org/10.1111/jiec.12921

Supporting Information S1

This supporting information contains detailed descriptions of the "sustainability aspects" we considered within the inclusion criteria for our literature search protocol. These "sustainability aspects" also served as coding categories for content analysis of our literature sample. Also included in this document are detailed descriptions of our framework components (*i.e.*, medical activities, "supporting" activities, and "background product systems"), which served as additional coding categories.

Supporting Information S2

This supporting information lists all of the literature searches we conducted between September 29 and November 17, 2017, and describes each in terms of the database searched and the keyword search phrase used.

Supporting Information S3

This supporting information describes our literature "sampling frame" of 1,748 articles in terms of the distribution of publications over time and between the subject matter classifications in the Scopus database.

Supporting Information S4

This supporting information provides a detailed analysis of the sustainability aspects (as described in Supporting Information S1) addressed in relation to categories of medical activities (table S1), supporting activities (table S2), and background product systems (table S3).

Supporting Information S5

This supporting information provides a complete list of the 157 articles analyzed, with bibliographic information and coding.

Appendix B

Supplementary content for Chapter 3

Supplementary content

1. Calculations for confidence intervals
2. Sample size estimation, with hypothetical worked example
3. Mathematical proof of equivalent O-LCA results using physical or economic values
4. Additional supplementary files available on the web

1. Calculations for confidence intervals

The margin of error for the estimated average environmental intensity of a "population" of *N* products is given by:

$$e = t_{n-1}^{*} \frac{s}{\sqrt{n}} \sqrt{\frac{N-n}{N-1}}$$

Where:

> *e* = margin of error for average environmental intensity (per unit of expenditure, for a given environmental impact category)
>
> t^*_{n-1} = critical value of *Student's t* distribution, with *n*-1 degrees of freedom, for a specified confidence level
>
> *s* = (sample) standard deviation of environmental intensity
>
> *n* = sample size (*i.e.*, number of unique products in the sample)
>
> *N* = population size (*i.e.*, total number of unique products purchased by the reporting organization during the reference period)

2. Sample size estimation, with hypothetical worked example

Using basic algebra, the equation for calculating the margin of error can be solved for the minimum sample size required to achieve a targeted margin of error (at a specified confidence level):

$$n = \frac{n_0 N}{n_0 + (N - 1)}$$

Where:

n_0 = sample size without finite population correction, calculated as $n_0 = \frac{{t^*_{n-1}}^2 s^2}{e^2}$

e = margin of error for average environmental intensity (per unit of expenditure, for a given environmental impact category)

t^*_{n-1} = critical value of *Student's t* distribution, with n-1 degrees of freedom, for a specified confidence level

s = (sample) standard deviation of environmental intensity

n = sample size with finite population correction (applicable when the sample size is "large" relative to the population size; 10% or more, following the rule of thumb suggested by DeVeaux et al. (2015))

N = population size

The problem, however, is that some of the values needed to calculate n_0, and therefore n, are unknown. The sample standard deviation (s) cannot be known without the sample data. The critical value, $t^*_{n\text{-}1}$, is also unknown, because it depends in part on the degrees of freedom (*i.e.*, n-1), which in turn depends on n. Therefore, these values need to be approximated; s can be approximated by using an educated guess or a very small "pilot" sample, and $t^*_{n\text{-}1}$ can be approximated using the corresponding critical value for a normal distribution. Once an initial estimate for n is calculated, the estimate can be recalculated using the corresponding value of $t^*_{n\text{-}1}$. As a hypothetical example, suppose a reporting organization purchased 1,500 unique products during the reference period (*i.e.*, N = 1,500), at a total cost of $1,500,000. Further assume that a pilot sample of 10 products has a standard deviation (s) of 0.5 kg CO_2 eq. per dollar spent. To achieve a 50% margin of error, on an estimated total GWP of 500 t CO_2 eq. (calculated based on the average environmental intensity estimated from the pilot sample of 10 products), with 95% confidence:

$$n_0 \approx \frac{(1.96)^2(0.5)^2}{(0.1667)^2} = 34.6 \rightarrow 35$$

Where:

$t^*_{n\text{-}1}$ is approximated by the critical value (at the 95% confidence level) for a normal distribution ≈ 1.96

s = 0.5 kg CO_2 eq./$

e = targeted margin of error for average environmental intensity = 0.1667 kg CO_2 eq./$ (*i.e.*, to achieve a 50% margin of error on the estimated total of 500 t CO_2 eq.)

$$n_0 = \frac{(2.037)^2(0.5)^2}{(0.1667)^2} = 37.3 \rightarrow 38$$

Where:

$t^*_{n\text{-}1} \approx 2.037$ for initial estimate of $n_0 = 35$

$s = 0.5$ kg CO_2 eq./$

e = targeted margin of error for average environmental intensity = 0.1667 kg CO_2 eq./$

As in this example, it is prudent to always round up when estimating the required sample size. Also, in this example, the finite population correction is not necessary given that the estimated sample size (*i.e.*, 38 unique products) is only 2.5% of the population size (*i.e.*, 1,500 unique products).

3. Mathematical proof of equivalent O-LCA results using physical or economic values

For simplicity, assume a reporting organization purchases two products: Product A and Product B.

The total quantity of Product A purchased, on a mass basis, is given by $q_A m_A$, where:

q_A = number of units of Product A purchased

m_A = mass per unit of Product A

Similarly, for Product B, the total quantity purchased, on a mass basis, is given by $q_B m_B$.

The total quantity of Product A purchased, on a cost basis, is given by $q_A c_A$, where:

q_A = number of units of Product A purchased

c_A = cost per unit of Product A

Similarly, for Product B, the total quantity purchased, on a cost basis, is given by $q_B c_B$.

Let the environmental impact per unit of Product A be denoted as e_A. Similarly, let the environmental impact per unit of Product B be denoted as e_B.

The environmental intensity, on a mass basis, of Product A is given by $\frac{e_A}{m_A}$. The environmental intensity, on a cost basis, of Product A is given by $\frac{e_A}{c_A}$. Similarly, the environmental intensity, on a mass basis, of Product B is given by $\frac{e_B}{m_B}$, and the environmental intensity, on a cost basis, of Product B is given by $\frac{e_B}{c_B}$.

The total environmental impact, for all units of Product A purchased, is given in three equivalent ways:

1. By number of units purchased: $q_A e_A$
2. By mass of units purchased: $q_A m_A \frac{e_A}{m_A} = q_A e_A$
3. By cost of units purchased: $q_A c_A \frac{e_A}{c_A} = q_A e_A$

Similarly, for Product B, $q_B m_B \frac{e_B}{m_B} = q_B c_B \frac{e_B}{c_B} = q_B e_B$.

The weighted average environmental intensity, on a mass basis, of Products A and B is given by:

$$\left(\frac{q_A m_A}{q_A m_A + q_B m_B}\right)\left(\frac{e_A}{m_A}\right) + \left(\frac{q_B m_B}{q_A m_A + q_B m_B}\right)\left(\frac{e_B}{m_B}\right) = \frac{q_A e_A + q_B e_B}{q_A m_A + q_B m_B}$$

Multiplying by the total mass of Products A and B purchased gives:

$$\frac{q_A e_A + q_B e_B}{q_A m_A + q_B m_B}(q_A m_A + q_B m_B) = q_A e_A + q_B e_B$$

Similarly, on a cost basis,

$$\left(\frac{q_A c_A}{q_A c_A + q_B c_B}\right)\left(\frac{e_A}{c_A}\right) + \left(\frac{q_B c_B}{q_A c_A + q_B c_B}\right)\left(\frac{e_B}{c_B}\right) = \frac{q_A e_A + q_B e_B}{q_A c_A + q_B c_B}$$

$$\frac{q_A e_A + q_B e_B}{q_A c_A + q_B c_B}(q_A c_A + q_B c_B) = q_A e_A + q_B e_B$$

4. Additional supplementary files available on the web

Additional supplementary files are available at https://uwaterloo.ca/industrial-ecology/cimprich-phd.

Appendix C

Supplementary content for Chapter 4

Recognizing that our studied hospital is located in the Canadian province of British Columbia, with a relatively low-carbon electrical grid (Canada Energy Regulator, 2021a), we consider how our results could change in the hypothetical scenario of an identical hospital in Alberta – a neighbouring province with a higher-carbon grid (Canada Energy Regulator, 2021b). In this scenario, the background process for the hospital's electricity use is changed from "market for electricity, low voltage – Canada, British Columbia" to "market for electricity, low voltage – Canada, Alberta" (both processes from the Ecoinvent 3.4 database). Additionally, the background process for water supply is changed from "market for tap water – Canada, Quebec" (as the closest proxy for British Columbia in the Ecoinvent 3.4 database) to "market group for tap water – Global" (given the lack of closer proxy for Alberta in the Ecoinvent 3.4 database). Finally, the background process for wastewater treatment is changed from "market for wastewater, average – Switzerland" (as the closest proxy for British Columbia in the Ecoinvent 3.4 database) to "market for wastewater, average – Rest of World" (given the lack of closer proxy for Alberta in the Ecoinvent 3.4 database). This scenario more than doubles the GWP contribution of the hospital's energy and water use (from 2,200 to 5,200 t CO_2 eq.), and therefore increases the hospital's total GWP to 6,600-8,000 t CO_2 eq., with 95% confidence.

Additional supplementary files available on the web

Additional supplementary files are available at https://uwaterloo.ca/industrial-ecology/cimprich-phd.

Appendix D

Re-expression of hospital O-LCA results

This appendix presents the results of the hospital O-LCA (Chapter 4) re-expressed in relation to Scopes 1, 2, and 3 per the *Greenhouse Gas Protocol Corporate Accounting and Reporting Standard* (WRI & WBCSD, 2004):

Scope 1: Direct emissions and resource use

Scope 2: Indirect emissions and resource use from purchased electricity

Scope 3: Other indirect emissions and resource use

See tables starting on the following page. Consistent with previous “top-down” studies of healthcare on a national-sector-level (Eckelman et al., 2018, 2020; Eckelman & Sherman, 2016; Malik et al., 2018; Nansai et al., 2020; Tennison et al., 2021; Weisz et al., 2020; Wu, 2019), Scope 3 dominates across all of the environmental indicators in the TRACI 2.1 impact assessment method used in the hospital O-LCA.

Table B1: Global warming potential

Process	Amount t CO_2 eq.	95% Confidence Lower Bound t CO_2 eq.	95% Confidence Upper Bound t CO_2 eq.
TOTAL	4.3E+03	3.5E+03	5.0E+03
Scope 3	**2.5E+03**	**1.8E+03**	**3.2E+03**
Supply-Chains	**8.7E+02**	**1.2E+02**	**1.6E+03**
Medical products	3.6E+02	4.2E+01	6.8E+02
Chemicals	2.3E+02	6.0E+01	3.9E+02
Non-drug pharmacy supplies	8.7E+01	1.9E+01	1.5E+02
Miscellaneous items A	6.9E+01	1.4E+01	1.2E+02
Laboratory supplies	5.5E+01	-9.1E+00	1.2E+02
Cleaning supplies	1.7E+01	1.2E+01	2.3E+01
Sterilization supplies	1.7E+01	3.7E+00	3.0E+01
Miscellaneous items C	1.6E+01	-2.2E+01	5.3E+01
Office supplies	1.1E+01	5.8E+00	1.7E+01
Computer and electronic supplies	8.4E+00	-4.5E-01	1.7E+01
Non-fuel heating supplies	1.8E+00	5.2E-01	3.1E+00
Bedding supplies	1.8E+00	-7.2E-01	4.3E+00
Hazardous waste containers	1.0E+00	1.0E+00	1.0E+00
Clinical nutrition	7.2E-02	1.2E-02	1.3E-01
Miscellaneous items B	4.8E-02	4.8E-02	4.8E-02
Hardware supplies	1.3E-02	-9.8E-03	3.6E-02
Water and Sewer	**6.3E+02**	**6.3E+02**	**6.3E+02**
83101506-01: Water treatment services, wastewater, average - CH	4.1E+02	4.1E+02	4.1E+02
83101501: Supply of water - CA-QC	2.2E+02	2.2E+02	2.2E+02
Pharmaceuticals	**4.8E+02**	**4.8E+02**	**4.8E+02**
Natural Gas, Scope 3	**2.1E+02**	**2.1E+02**	**2.1E+02**
Patient Diets	**1.8E+02**	**1.8E+02**	**1.8E+02**
Delivery, Freight & Courier	**5.9E+01**	**5.9E+01**	**5.9E+01**
Waste Streams	**3.6E+01**	**3.6E+01**	**3.6E+01**
Housekeeping, Laundry & Linen Services	**2.0E+01**	**2.0E+01**	**2.0E+01**
Fuel Oil, Scope 3	**6.4E+00**	**6.4E+00**	**6.4E+00**
Patient Transportation	**5.1E+00**	**5.1E+00**	**5.1E+00**
Scope 1	**1.4E+03**	**1.4E+03**	**1.4E+03**
Natural Gas, Scope 1	**9.8E+02**	**9.8E+02**	**9.8E+02**
Anesthetic Gas Emissions	**4.1E+02**	**4.1E+02**	**4.1E+02**
Fuel Oil, Scope 1	**4.1E+01**	**4.1E+01**	**4.1E+01**
Scope 2	**3.2E+02**	**3.2E+02**	**3.2E+02**

Table B2: Acidification potential

Process	Amount (kg SO_2 eq.)
TOTAL	1.2E+04
Scope 3	**9.9E+03**
Water and Sewer	**4.6E+03**
83101506-01: Water treatment services, wastewater, average - CH	3.6E+03
83101501: Supply of water - CA-QC	1.0E+03
Supply-Chains	**4.0E+03**
Medical products	1.5E+03
Chemicals	9.9E+02
Miscellaneous items A	4.3E+02
Non-drug pharmacy supplies	3.3E+02
Miscellaneous items C	2.5E+02
Laboratory supplies	2.4E+02
Sterilization supplies	7.9E+01
Cleaning supplies	7.6E+01
Computer and electronic supplies	6.7E+01
Office supplies	5.9E+01
Non-fuel heating supplies	1.6E+01
Bedding supplies	8.5E+00
Hazardous waste containers	3.7E+00
Clinical nutrition	2.7E-01
Miscellaneous items B	1.8E-01
Hardware supplies	6.4E-02
Natural Gas, Scope 3	**6.8E+02**
Delivery, Freight & Courier	**3.0E+02**
Housekeeping, Laundry & Linen Services	**9.0E+01**
Waste Streams	**8.1E+01**
Fuel Oil, Scope 3	**6.5E+01**
Patient Transportation	**2.0E+01**
Scope 2	**9.9E+02**
Scope 1	**9.4E+02**
Fuel Oil, Scope 1	**4.9E+02**
Natural Gas, Scope 1	**4.5E+02**

Table B3: Human toxicity, carcinogenic effects

Process	Amount (CTUh)
TOTAL	5.6E-01
Scope 3	**5.2E-01**
Water and Sewer	**3.2E-01**
83101501: Supply of water - CA-QC	1.7E-01
83101506-01: Water treatment services, wastewater, average - CH	1.6E-01
Supply-Chains	**1.8E-01**
Chemicals	1.3E-01
Medical products	2.3E-02
Miscellaneous items A	1.1E-02
Laboratory supplies	4.6E-03
Miscellaneous items C	4.6E-03
Non-drug pharmacy supplies	3.2E-03
Computer and electronic supplies	2.0E-03
Office supplies	1.8E-03
Sterilization supplies	1.3E-03
Cleaning supplies	7.0E-04
Non-fuel heating supplies	6.9E-04
Bedding supplies	1.6E-04
Hazardous waste containers	3.8E-05
Hardware supplies	6.4E-06
Clinical nutrition	4.2E-06
Miscellaneous items B	1.8E-06
Natural Gas, Scope 3	**5.9E-03**
Waste Streams	**2.4E-03**
Delivery, Freight & Courier	**1.9E-03**
Housekeeping, Laundry & Linen Services	**1.7E-03**
Patient Transportation	**3.9E-04**
Fuel Oil, Scope 3	**2.3E-04**
Scope 2	**3.9E-02**
Scope 1	**6.7E-03**
Natural Gas, Scope 1	**6.7E-03**
Fuel Oil, Scope 1	**5.7E-05**

Table B4: Ecotoxicity

Process	Amount (CTUe)
TOTAL	6.3E+07
Scope 3	**3.6E+07**
Supply-Chains	**1.9E+07**
Chemicals	6.0E+06
Miscellaneous items A	5.1E+06
Medical products	3.9E+06
Miscellaneous items C	2.1E+06
Computer and electronic supplies	8.1E+05
Office supplies	4.3E+05
Laboratory supplies	4.0E+05
Non-drug pharmacy supplies	3.4E+05
Non-fuel heating supplies	1.3E+05
Sterilization supplies	1.3E+05
Cleaning supplies	8.7E+04
Bedding supplies	2.4E+04
Hazardous waste containers	3.6E+03
Clinical nutrition	3.5E+02
Hardware supplies	2.8E+02
Miscellaneous items B	1.8E+02
Water and Sewer	**1.5E+07**
83101506-01: Water treatment services, wastewater, average - CH	1.1E+07
83101501: Supply of water - CA-QC	4.1E+06
Natural Gas, Scope 3	**4.5E+05**
Delivery, Freight & Courier	**3.4E+05**
Waste Streams	**2.7E+05**
Housekeeping, Laundry & Linen Services	**2.4E+05**
Patient Transportation	**1.9E+05**
Fuel Oil, Scope 3	**2.0E+04**
Scope 2	**2.6E+07**
Scope 1	**1.1E+06**
Natural Gas, Scope 1	**1.1E+06**
Fuel Oil, Scope 1	**2.5E+04**

Table B5: Eutrophication potential

Process	Amount (kg N eq.)
TOTAL	3.9E+04
Scope 3	**3.6E+04**
Water and Sewer	**3.2E+04**
83101506-01: Water treatment services, wastewater, average - CH	3.1E+04
83101501: Supply of water - CA-QC	6.1E+02
Supply-Chains	**3.9E+03**
Chemicals	1.1E+03
Miscellaneous items A	9.5E+02
Medical products	9.2E+02
Miscellaneous items C	4.0E+02
Non-drug pharmacy supplies	1.4E+02
Computer and electronic supplies	1.3E+02
Laboratory supplies	1.0E+02
Office supplies	8.0E+01
Sterilization supplies	4.4E+01
Cleaning supplies	3.7E+01
Non-fuel heating supplies	2.4E+01
Bedding supplies	7.2E+00
Hazardous waste containers	1.7E+00
Clinical nutrition	9.3E-02
Miscellaneous items B	7.8E-02
Hardware supplies	4.6E-02
Waste Streams	**1.4E+02**
Natural Gas, Scope 3	**1.1E+02**
Housekeeping, Laundry & Linen Services	**7.5E+01**
Delivery, Freight & Courier	**6.8E+01**
Fuel Oil, Scope 3	**2.3E+01**
Patient Transportation	**1.1E+01**
Scope 2	**2.4E+03**
Scope 1	**2.9E+02**
Natural Gas, Scope 1	**2.9E+02**
Fuel Oil, Scope 1	**3.2E+00**

Table B6: Fossil fuel depletion

Process	Amount (MJ surplus)
TOTAL	5.7E+06
Scope 3	**5.2E+06**
Natural Gas, Scope 3	**2.8E+06**
Supply-Chains	**1.7E+06**
Medical products	7.8E+05
Non-drug pharmacy supplies	2.8E+05
Laboratory supplies	2.0E+05
Chemicals	1.6E+05
Miscellaneous items A	8.9E+04
Cleaning supplies	4.8E+04
Sterilization supplies	3.5E+04
Miscellaneous items C	2.3E+04
Office supplies	1.5E+04
Computer and electronic supplies	9.2E+03
Hazardous waste containers	3.5E+03
Bedding supplies	1.5E+03
Non-fuel heating supplies	1.5E+03
Clinical nutrition	1.9E+02
Miscellaneous items B	1.6E+02
Hardware supplies	1.2E+01
Water and Sewer	**5.1E+05**
83101506-01: Water treatment services, wastewater, average - CH	3.2E+05
83101501: Supply of water - CA-QC	1.9E+05
Delivery, Freight & Courier	**1.3E+05**
Fuel Oil, Scope 3	**1.0E+05**
Housekeeping, Laundry & Linen Services	**2.7E+04**
Patient Transportation	**9.7E+03**
Waste Streams	**9.3E+03**
Scope 2	**3.5E+05**
Scope 1	**1.2E+05**
Natural Gas, Scope 1	**1.2E+05**
Fuel Oil, Scope 1	**-2.9E+01**

Table B7: Human toxicity, non-carcinogenic effects

Process	Amount (CTUh)
TOTAL	3.3E+00
Scope 3	**3.1E+00**
Water and Sewer	**2.5E+00**
83101506-01: Water treatment services, wastewater, average - CH	2.3E+00
83101501: Supply of water - CA-QC	2.0E-01
Supply-Chains	**6.3E-01**
Chemicals	1.9E-01
Miscellaneous items A	1.9E-01
Miscellaneous items C	9.7E-02
Medical products	6.9E-02
Computer and electronic supplies	3.3E-02
Office supplies	1.5E-02
Laboratory supplies	1.4E-02
Non-drug pharmacy supplies	9.8E-03
Non-fuel heating supplies	5.9E-03
Sterilization supplies	5.3E-03
Cleaning supplies	3.0E-03
Bedding supplies	4.2E-04
Hazardous waste containers	9.0E-05
Clinical nutrition	1.3E-05
Hardware supplies	7.7E-06
Miscellaneous items B	5.1E-06
Natural Gas, Scope 3	**1.7E-02**
Delivery, Freight & Courier	**1.4E-02**
Waste Streams	**6.8E-03**
Housekeeping, Laundry & Linen Services	**4.6E-03**
Patient Transportation	**1.8E-03**
Fuel Oil, Scope 3	**8.2E-04**
Scope 2	**1.8E-01**
Scope 1	**2.8E-02**
Natural Gas, Scope 1	**2.5E-02**
Fuel Oil, Scope 1	**2.4E-03**

Table B8: Ozone depletion potential

Process	Amount (kg CFC-11 eq.)
TOTAL	6.2E-01
Scope 3	**5.7E-01**
Supply-Chains	**3.0E-01**
Medical products	2.4E-01
Chemicals	1.5E-02
Laboratory supplies	1.0E-02
Miscellaneous items A	8.0E-03
Non-drug pharmacy supplies	7.0E-03
Miscellaneous items C	4.2E-03
Bedding supplies	3.0E-03
Sterilization supplies	2.4E-03
Cleaning supplies	2.3E-03
Non-fuel heating supplies	1.2E-03
Office supplies	1.2E-03
Computer and electronic supplies	1.1E-03
Hazardous waste containers	4.9E-05
Clinical nutrition	4.1E-06
Miscellaneous items B	3.5E-06
Hardware supplies	9.5E-07
Natural Gas, Scope 3	**1.5E-01**
Water and Sewer	**6.4E-02**
83101506-01: Water treatment services, wastewater, average - CH	4.1E-02
83101501: Supply of water - CA-QC	2.3E-02
Housekeeping, Laundry & Linen Services	**2.7E-02**
Delivery, Freight & Courier	**1.5E-02**
Fuel Oil, Scope 3	**1.2E-02**
Waste Streams	**1.5E-03**
Patient Transportation	**1.1E-03**
Scope 2	**3.0E-02**
Scope 1	**2.7E-02**
Natural Gas, Scope 1	**2.7E-02**
Fuel Oil, Scope 1	**9.2E-07**

Table B9: Human toxicity, respiratory effects

Process	**Amount (kg $PM_{2.5}$ eq.)**
TOTAL	2.9E+03
Scope 3	**2.1E+03**
Supply-Chains	**8.9E+02**
Chemicals	3.3E+02
Medical products	2.9E+02
Miscellaneous items A	8.2E+01
Non-drug pharmacy supplies	5.4E+01
Laboratory supplies	4.3E+01
Miscellaneous items C	3.5E+01
Sterilization supplies	1.6E+01
Office supplies	1.3E+01
Cleaning supplies	1.3E+01
Computer and electronic supplies	1.2E+01
Non-fuel heating supplies	2.8E+00
Bedding supplies	2.2E+00
Hazardous waste containers	6.2E-01
Clinical nutrition	3.5E-02
Miscellaneous items B	3.0E-02
Hardware supplies	2.8E-02
Water and Sewer	**7.1E+02**
83101506-01: Water treatment services, wastewater, average - CH	4.3E+02
83101501: Supply of water - CA-QC	2.8E+02
Pharmaceuticals	**3.3E+02**
Natural Gas, Scope 3	**5.6E+01**
Delivery, Freight & Courier	**4.0E+01**
Housekeeping, Laundry & Linen Services	**2.1E+01**
Waste Streams	**1.3E+01**
Fuel Oil, Scope 3	**6.6E+00**
Patient Transportation	**3.0E+00**
Scope 2	**7.0E+02**
Scope 1	**1.4E+02**
Natural Gas, Scope 1	**9.3E+01**
Fuel Oil, Scope 1	**4.7E+01**

Table B10: Smog formation potential

Process	Amount (kg O_3 eq.)
TOTAL	1.6E+05
Scope 3	**1.4E+05**
Supply-Chains	**6.9E+04**
Medical products	3.8E+04
Chemicals	1.2E+04
Non-drug pharmacy supplies	5.1E+03
Miscellaneous items A	4.3E+03
Laboratory supplies	4.1E+03
Miscellaneous items C	1.3E+03
Sterilization supplies	1.2E+03
Cleaning supplies	1.1E+03
Office supplies	7.3E+02
Computer and electronic supplies	5.7E+02
Non-fuel heating supplies	1.1E+02
Bedding supplies	9.3E+01
Hazardous waste containers	4.6E+01
Clinical nutrition	4.6E+00
Miscellaneous items B	2.7E+00
Hardware supplies	7.2E-01
Water and Sewer	**5.1E+04**
83101506-01: Water treatment services, wastewater, average - CH	3.7E+04
83101501: Supply of water - CA-QC	1.4E+04
Natural Gas, Scope 3	**7.7E+03**
Delivery, Freight & Courier	**7.6E+03**
Housekeeping, Laundry & Linen Services	**9.6E+02**
Waste Streams	**7.5E+02**
Fuel Oil, Scope 3	**5.5E+02**
Patient Transportation	**3.7E+02**
Scope 2	**1.2E+04**
Scope 1	**1.2E+04**
Natural Gas, Scope 1	**1.0E+04**
Fuel Oil, Scope 1	**1.8E+03**