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Life cycle sustainability assessment framework for water sector resource recovery solutions: Strengths and weaknesses

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A R T I C L E I N F O

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A B S T R A C T

Resource recovery solutions are an essential part of a sustainable water sector. Sustainability of these solutions needs to be analysed to assess, compare and optimize them. Life Cycle Sustainability Assessment (LCSA) is the most commonly used framework for sustainability assessment. This review paper discusses three critical characteristics of water sector resource recovery solutions: (i) their potential to actively benefit natural processes through reciprocal services, (ii) their dependence upon natural resources and processes, and (iii) their goal to avoid transgression of environmental thresholds. We analyse these three characteristics in the context of the following features of LCSA: (i) it being a damage assessment-based framework, (ii) its treatment of economic and natural capital as substitutable and (iii) the absence of environmental thresholds and past emissions in its environmental assessment methodology. We use a real-life resource recovery case study from the Netherlands to evaluate and demonstrate the mentioned features of the existing LCSA framework. Our review indicates that, LCSA can be modified for application to resource recovery solutions if it includes reciprocity towards nature as an essential component, limits compensations between economic welfare and environmental damage, and incorporates environmental thresholds and past emissions.

Important terms

Economic welfare: It is defined as the sum of producer and consumer surplus. Producer surplus refers to a situation where consumers are willing to pay more for a product than the minimum amount a producer is willing to accept for it. Production cost reduction is considered here as an improvement of economic welfare for the producer.

Industrial symbiosis: It is a concept from industrial ecology concerned with resource and energy optimization based on linkages between firms. This linkage increases the efficiency of the entire cluster of energy/material flows involved in the link (Li, 2017).

Natural capital: It is the whole natural environment, defined by the stocks of natural assets like forests, water, flora, fauna, atmosphere, etc., which provide humans with useful flows of goods and services (Daly and Goodland, 1996).

Normative values: Values essentially understood to be justified as something we ought to have (Lindenberg and Steg, 2007). The opposite is a descriptive value that only describes the state of how something 'is', whereas normativity points to how something 'ought to be.'

Planetary boundary: These are thresholds within which humanity can safely operate. Transgressing these can be deleterious or catastrophic (Rockström et al., 2009).

System: It is a set of elements interacting with one another purposefully to achieve a common goal. For example, the wastewater treatment system comprises several elements like primary settler, sludge digester, etc., that all work to achieve clean water.

Thresholds: Intrinsic characteristics of a human-environmental system defined by a control variable value beyond which the functioning of this system is non-linearly transformed or disrupted (Rockström et al., 2009).

1. Introduction

As a society desirous of comfort and prosperity in the face of limited natural capital, sustainability is often presented as a solution. Although sustainability is not a well-defined concept and remains open to various interpretations subject to context (Purvis et al., 2019; Wulf et al., 2019), there is some agreement on its three pillars, namely the environment, society, and economy (Florindo et al., 2020; Godskesen et al., 2018; Purvis et al., 2019).

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Resource recovery and reuse is one way to promote sustainability of the water sector (Wang et al., 2015). Technological solutions for resource recovery exist, but there is a need for planning and designing methodologies for selecting the most sustainable options (Puchongkawarin et al., 2015; Van Der Hoek et al., 2016). Decision-makers have to assess and ensure the sustainability of proposed technological solutions and can benefit from an assessment framework. A framework for sustainability assessment must inform decision-makers on its three pillars and not just in isolation but in an integrated and inter-disciplinary manner.

The predominant framework used for the three-pillar sustainability assessment in an integrated and inter-disciplinary manner is called Life Cycle Sustainability Assessment (LCSA) (Gloria et al., 2017). LCSA evaluates environmental impacts using Life Cycle Assessment (LCA), economic costs using Life Cycle Costing (LCC), and social impacts using a Social-Life Cycle Assessment (S-LCA) (Guinée et al., 2011). In this paper, the focus will be on the environmental and economic assessment of sustainability. This is because methods used for S-LCA are fragmented, lack a general theoretical basis or standards (Haase et al., 2020; Taelman et al., 2020), and developing them is beyond the scope of this paper.

The objective of our review is to analyse the strengths and weaknesses of the existing LCSA framework, in case that it is used to assess the sustainability of water sector resource recovery solutions and to suggest modifications, if required. We analyse the framework using literature on LCSA application to resource recovery solutions and we evaluate it by applying it on a real-life case study.

The review paper is organised as follows. It begins with a section introducing resource recovery solutions and LCSA. A brief description of characteristics of water sector resource recovery solutions are presented in Strengths and weaknesses of LCSA are described in Section 2.2. Section 3 is the methods section that starts with a short description of a water sector resource recovery case study. Sections 3.2, 3.3, and 3.4 introduce LCA, LCC and MCDA respectively as methods used in our case study evaluation. Section 4 presents results obtained in the case study evaluation for each of the three methods. This is followed by a discussion of the characteristics of LCSA and suggested modifications using the case study results as the basis in Section 5. Based on the demonstrated characteristics and suggested modifications to LCSA, Section 6 examines future research needs, and conclusions follow in Section 7. Supplementary material 1 (S1) includes the LCA template report based on ISO 14,040. S2 contains spreadsheets relevant for LCA and LCC inputs and results.

2. Resource recovery solutions and life cycle sustainability assessment

2.1. Water sector resource recovery solutions

Recovering resources from water is an essential step in making water management fit the circular economy paradigm and become sustainable. This section briefly discusses three critical characteristics of water sector resource recovery solutions from literature.

Firstly, recovered resources from water management can positively impact ecosystems, thereby fostering a reciprocal flow of benefits between society and nature (Trimmer et al., 2019). For example, sewage sludge application to soil can achieve erosion control, improvement of soil structure, and better quality vegetation (Bachev and Ivanov, 2021). The possibility of such positive impacts indicates that resource recovery solutions need not remain focussed on damage reduction and mitigation. Rather, they can also serve as contributors towards nature restoration.

Secondly, resource recovery solutions rely on nature. Generally speaking, any economic activity, including water sector resource recovery solutions, depends on natural resources and processes. For example, nutrient recovery from wastewater by adsorption makes use of the mineral zeolite (Vera-Puerto et al., 2020), a natural resource. Along similar lines, reuse of water for aquifer recharge, which has benefits such as land subsidence prevention, groundwater recharge, etc., depends on the natural filtering capacity of soils (Yuan et al., 2016). This exhibits that resource recovery solutions rely on the availability of natural resources and natural processes.

Thirdly, resource recovery solutions help manage resources better and avoid transgression of environmental thresholds such as planetary boundaries (Velenturf and Purnell, 2017). These solutions are motivated by the scarcity of natural resources (Chrispim et al., 2020). Therefore, preventing natural resource depletion and transgression of environmental thresholds are goals associated with resource recovery solutions.

2.2. Life cycle sustainability assessment of water sector resource recovery solutions

For appraising sustainability, LCSA uses Life Cycle Assessment, Life Cycle Costing, and Social Life Cycle Assessment methods (Costa et al., 2019). The contribution of LCSA is commendable in mainstreaming life cycle thinking and broadening the impacts considered to include environmental, economic, and social dimensions. But application examples of LCSA to resource recovery solutions are rare (Millward-Hopkins et al., 2018). Whether the framework is sufficient for assessing the sustainability of water sector resource recovery remains to be explored. In this section, there is a discussion on some strengths and weaknesses of LCSA. These are presented under three categories: conceptual, ontological, and methodological.

2.2.1. Conceptual characteristics

In an LCSA, the environmental assessment makes use of LCA to measure eco-efficiency. The strength of LCA lies in its comprehensive coverage of environmental impact categories and inclusion of entire life cycles, thus preventing burden shifting (Hauschild et al., 2017). But, it is noteworthy that the concept of eco-efficiency is centred on damage reduction (Barbiroli, 2010; Hauschild, 2015; Niero et al., 2017). Thus, a damage reduction approach is being used to assess solutions that may positively impact nature.

LCSA framework applications have yet to explicitly consider reciprocity from humans to nature as an essential component to the best of our knowledge. In fact, reciprocity can potentially fit well within the popular industrial symbiosis concept. Industrial symbiosis is inspired by the biological ecosystem's mutualistic interactions (Chatterjee et al., 2021). In a biological symbiotic system, an organism may benefit, get harmed, or remain unaffected through the association (Aydt et al., 2008). Yet, the spirit of developing industrial systems that mimic biological symbiosis lies in interactions that mutually benefit organisms and the environment.

2.2.2. Ontological characteristics

LCSA framework is remarkable for broadening the scope of analysis from only environmental to including social and economic dimensions (Guinée et al., 2011). It ensures a holistic assessment preventing neglect of adverse social or economic consequences. But, how to aggregate the results from the three sustainability dimensions is not clear (Dong and Ng, 2016). The aggregation procedure used is consequential.

Any economic activity, including resource recovery solutions, depends on natural resources and processes. Yet, most LCSA application studies seem to accept the idea of competing environmental and economic objectives. This is evident because studies aggregate environmental and economic indicators in a manner that allows compensations between them. In other words, low performance on the environmental criterion can be compensated for by high economic performance.

The most common approach of aggregating results of economic and environmental assessment is linear aggregation into a single indicator using methods such as Analytical Hierarchy Process (AHP) or Multi-Attribute Utility Theory (MAUT) (Wulf et al., 2019). For example, in Sun et al. (2020), the authors use linear additive aggregation for weighted sustainability criteria: eutrophication, carbon intensity, resilience, and cost for assessing the sustainability of wastewater management alternatives (they also used sensitivity analysis for different weighting schemes). This method helps incorporate different stakeholder perspectives into decision-making. But, there is a fundamental error in its use of trade-offs that falsely assumes that economic welfare (production cost reduction) and environmental damage (eutrophication, carbon intensity) are substitutable. The different weighting schemes only point towards different degrees of substitutability.

When we make a purchase, we give up a sum of money to acquire goods. Rationality dictates that we do this only when we feel that trading the money for the goods has not left us worse off. Trade-offs, thus, can be made between substitutes. When we treat environmental damage and economic welfare as substitutable, it effectively implies that economic capital is substitutable with natural resources/processes and that we can forego one of them for the other. There are three issues with this.

Firstly, our socio-economic system could not be built without natural resources and processes (Daly and Goodland, 1996; Glavič and Lukman, 2007). Nor can the functioning of our existing systems continue indefinitely without infinite energy. While practically unlimited energy is available from the sun, to capture it, infrastructure is needed, building which, requires natural resources. Therefore, economic welfare cannot be traded with natural capital since the former depends upon the latter. In case, natural capital is depleted, we effectively deplete the means to generate economic welfare. A weak sustainability notion suggests that human-made capital can replace natural capital. We contend that the burden of proof must lie with the weak sustainability proponent. This is because, so far, the weak sustainability notion has mostly been found invalid (Biely et al., 2018; Lindmark et al., 2018; Qasim et al., 2020; Shang et al., 2019).

Secondly, natural capital is distinct from built capital in a way that precludes their comparison. The former is characterized by the presence of thresholds beyond which damages are irreversible. Species extinction, for example, is irreversible, i.e., a lost species and its consequences on the overall ecosystem functioning cannot be repaired or undone. Natural processes have thresholds; for example, the auto-purification process of water gets overloaded above a certain pollutant concentration beyond which the process is disrupted (Pelenc and Ballet, 2015). Whereas built capital is never irrevocably lost as long as natural resources and processes are available.

Lastly, natural capital fulfils many functions such as production, pollution absorption, etc. (Ekins, 2014). Built capital is usually meant to perform a single anthropocentric function. If built capital were to replace natural capital, many non-anthropocentric but crucial functions would no longer be fulfilled.

2.2.3. Methodological characteristics

LCSA framework successfully guides eco-efficiency improvements, but it does not determine if a service or product is sustainable in absolute terms (Hauschild et al., 2017).

In LCSA, LCA is almost exclusively used to measure eco-efficiency, i.e., how much resource depletion or emission is caused by a unit operation (Hauschild et al., 2017; Pelletier et al., 2019). However, equating eco-efficiency with sustainability without considering past emissions and environmental thresholds leads to a context-less assessment. A few examples of how environmental sustainability is being equated with eco-efficiency are discussed below.

Canaj et al. (2021) present how wastewater reuse for irrigation may have lower negative environmental impacts on human health and ecosystems damage. They use the term environmental benefit to refer to reduced environmental damage. Sun et al. (2020) compare four alternative wastewater treatment pathways concerning carbon emission and eutrophication intensity. They discover lower carbon emission and eutrophication for decentralized and centralized-decentralized hybrid wastewater treatment plants compared to centralized systems. Due to lower emissions, they point to the decentralized systems as being more sustainable. For quantifying the environmental sustainability of resource recovery systems from water, Cornejo et al. (2019) suggest using environmental metrics related to process inputs, recovered products, wastes, and emissions, amongst others, to assess sustainability. None of the mentioned studies made use of environmental thresholds for contextualizing eco-efficiency. Thus, reducing used resources, wastes, and emissions, is equated with environmental sustainability. These studies provide two observations. First, improving eco-efficiency has become the most widely accepted proposition to a sustainable future, as noted b (Sandberg et al., 2019). Second, LCA results are usually not linked to environmental thresholds or carrying capacities (Bjørn and Hauschild, 2015). Equating such context-less eco-efficiency with sustainability leads to two problems.

Firstly, lack of context regarding past emissions and environmental thresholds can lead to underestimation of the urgency of phenomena, such as climate change. For example, the build-up of carbon stocks in the atmosphere and its threshold value are ignored when results are only expressed in quantities of CO_2 emission.

Secondly, for emissions that may cause irreversible environmental damages, context-less eco-efficiency comparison of alternatives only determines which alternative can postpone threshold transgression longer. As an example, continuous phosphorus (P) discharge to water bodies can potentially lead to irreparable damage to the aquatic ecosystem (Biswas et al., 2017). An alternative with lower P discharge to water bodies does not necessarily translate to a sustainable solution unless the carrying capacity of the water body is respected. Thus, such emissions rate must be analysed in the context of the carrying capacity of the receiving environmental compartment.

It may be considered that with continuous eco-efficiency improvement, it is possible to postpone resource depletion and irreparable environmental damage indefinitely. But, continuous eco-efficiency improvement across all sectors is unlikely to keep up with rising affluence and population, as shown by Hauschild et al. (2017) using the IPAT equation. In this equation, three factors affecting negative environmental impacts (I) are recognised. These are Population (P), Affluence (A), and Technology (T) factors (Chertow, 2000). LCA targets the 'T' dimension (Hauschild et al., 2017).

Table 1 summarises the strengths and weaknesses of LCSA, and Section 3 will demonstrate these on a case study.

3. Methods

This section introduces methods used to evaluate the characteristics of LCSA summarized in Table 1. The evaluation will be based on a resource recovery case study belonging to the water sector.

Table 1
Strengths and weaknesses of Life Cycle Sustainability Assessment. This table shows which characteristics of LCSA can be modified to assess resource recovery solutions better.

Classification	Strengths and Weaknesses of LCSA and its Methods
Conceptual	LCSA avoids burden shifting by including a wide variety of impact categories and analysing complete life cycles. But, it remains focussed on environmental damage assessment and could benefit from including an environmental benefit assessment methodology.
Ontological	LCSA extends the scope of analysis to the three dimensions of environment, economy, and society. But, linear aggregation of results from these three dimensions allows complete compensation between environmental and economic capitals.
Methodological	LCSA enables eco-efficiency improvements. But, absolute sustainability cannot be judged solely with eco-efficiency measurements. Including current emissions and environmental thresholds can make the framework more comprehensive.

3.1. Case study description

In this section, we describe a water sector resource recovery solution under development that will be used as our case study. We choose the water sector because water is a basic necessity for society, and water management for human use is known to cause non-renewable resource depletion, high energy use, and harmful emissions (Morley et al., 2016). The water sector also provides several opportunities for resource recovery (Zhang et al., 2020). Some solutions are being designed to remedy environmental damages through resource recovery. One example is used as a case study.

A company in Amsterdam, called NPSP, is developing a bio-composite material using resources recovered from the managed urban water cycle. They use two types of recovered resources as raw materials, namely calcite and water reeds.

Calcium Carbonate ($CaCO_3$), commonly known as calcite, is used in the drinking water softening process in the Netherlands. The softening process consists of an upward-flow cylinder partly filled with calcite. Caustic soda is added to the hard water to raise its pH. This leads to a supersaturated condition in the water for calcium carbonate. As a result, calcium carbonate crystallizes around a seeding material. Sand can be used as seeding material. But it is also possible to reuse the crystallized calcite as seeding material and create a closed-loop recycling system (Schetters et al., 2015). Only a part of the calcite produced by the softening reactors needs to be reused as seeding material; the rest is available as raw-material for bio-composites. Water reeds, growing naturally along the canals and rivers in the Netherlands, are cut and collected regularly. They are cut down to 3–6 mm in length. These materials have to be glued together for which resins are used. NPSP is contemplating between unsaturated polyester resin and bio-based resins. We assume polyester resin in our model.

Calcite, reed fibres, and polyester resin are mixed to form a Bulk Moulding Compound (BMC), which is heat-pressed to make boards of standard size 600 mm x 600 mm x 10.5 mm. The boards are expected to be used as construction material for river/canal bank protection. Fig. 1 shows a flowchart describing the various processes of the bio-composite life-cycle.

3.2. Life cycle assessment (LCA)

An LCA is conducted on the material described in Section 3.1. The ISO 14040 template of LCA is followed, which consists of goal & scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation. We show only those portions of the LCA needed to understand its goal and scope and that are directly relevant to our discussion of LCSA. Supplementary Material 1 contains the complete LCA results.

3.2.1. Goal definition

The LCA will reveal environmental impacts associated with the production, use, and recycling of the NPSP bio-composite material and identify environmental hotspots. The study targets the bio-composite manufacturing company, water treatment facilities, and academics.

This LCA is being conducted primarily to analyse the use of the LCSA framework in water sector resource recovery solutions and demonstrate some of its characteristics. Hence, outcomes of the LCA should be seen as indicative. The actual impacts calculated can serve as a basis for a more detailed future analysis.

3.2.2. Scope definition

The economic system is highly complex. There is a large number of processes involved in manufacturing any product. The scope of an LCA has to be limited to critical processes, and thus, a system boundary is defined. The system here refers to all processes required to deliver the function of a product. It begins with raw material extraction and, after a canal bank lifespan of 25 years, ends with the recycling of its materials to make new bio-composites. Thus, this is a cradle to cradle LCA. Fig. 2 shows the system flowchart.

Through an LCA, environmental impacts from fulfilling a unit human demand are calculated, also known as the functional unit. In this study, the functional unit is 1000 mm of canal bank protection down to a water depth of 600 mm for 25 years. For this, a bio-composite board of length 1000 mm and width 10.5 mm is required. At every 1000 mm, a post of dimensions 100 mm x 100 mm x 2000 mm is installed. Thus, the total volume of material required per metre of canal bank protection is equivalent to 6.96 boards of standard size 600 mm x 600 mm x 10.5 mm, which is the reference flow.

3.2.3. Life cycle inventory

The life cycle inventory phase involves the compilation of elementary flow data, i.e., flows that pass between the system boundary and the natural environment. The elementary flow of biogenic CO_2 absorption by water reeds is of particular significance for our discussion. We present details of this process here.

Water-side reeds grow naturally along canal or river banks. Over their lifetime, they serve as a CO_2 stock (Zhou et al., 2009). Since the bio-composite is expected to be collected for recycling after use, the biogenic carbon stored is credited as negative emission. The negative emission credit is calculated as the embedded carbon content of reeds. Only above ground biomass of water reeds is considered, adding up to 1.459 kg/m^2 (Zhou et al., 2009). The net uptake of CO_2 by reeds is assumed to be 65±14 g C/m^2 (Zhou et al., 2009). Thus, net CO_2 accumulated in reeds is calculated as 46.42±10 g C/m^2. For the complete LCI, Supplementary Material S2.1 may be consulted.

3.3. Life cycle costing

Life Cycle Costing (LCC) is a methodology to estimate the total costs of a product/service over its lifetime (Gluch and Baumann, 2004). Costs of purchasing and transporting raw materials, raw material pre-treatment, manufacturing of bio-composite, installing at canal site, and material recovery by shredding are added to calculate the life cycle cost of the bio-composite. The Source of data is NPSP. The costs are only estimations, and strong conclusions regarding the actual LCC of the material are not recommended.

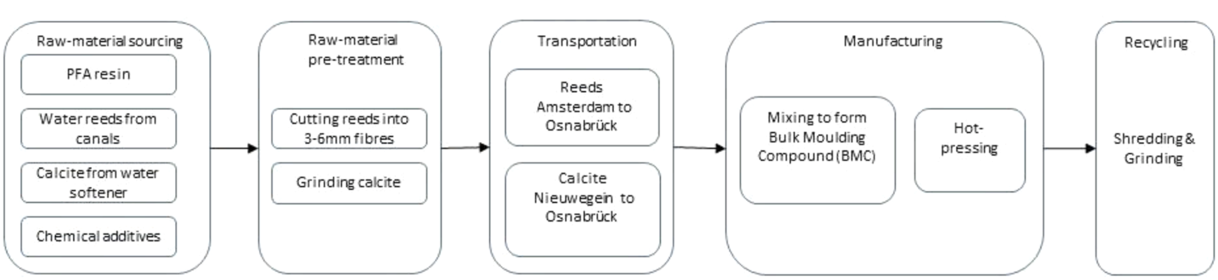

Fig. 1. Life cycle stages of NPSP bio-composite: Raw-material sourcing, raw material pre-treatment, transportation, manufacturing and recycling.

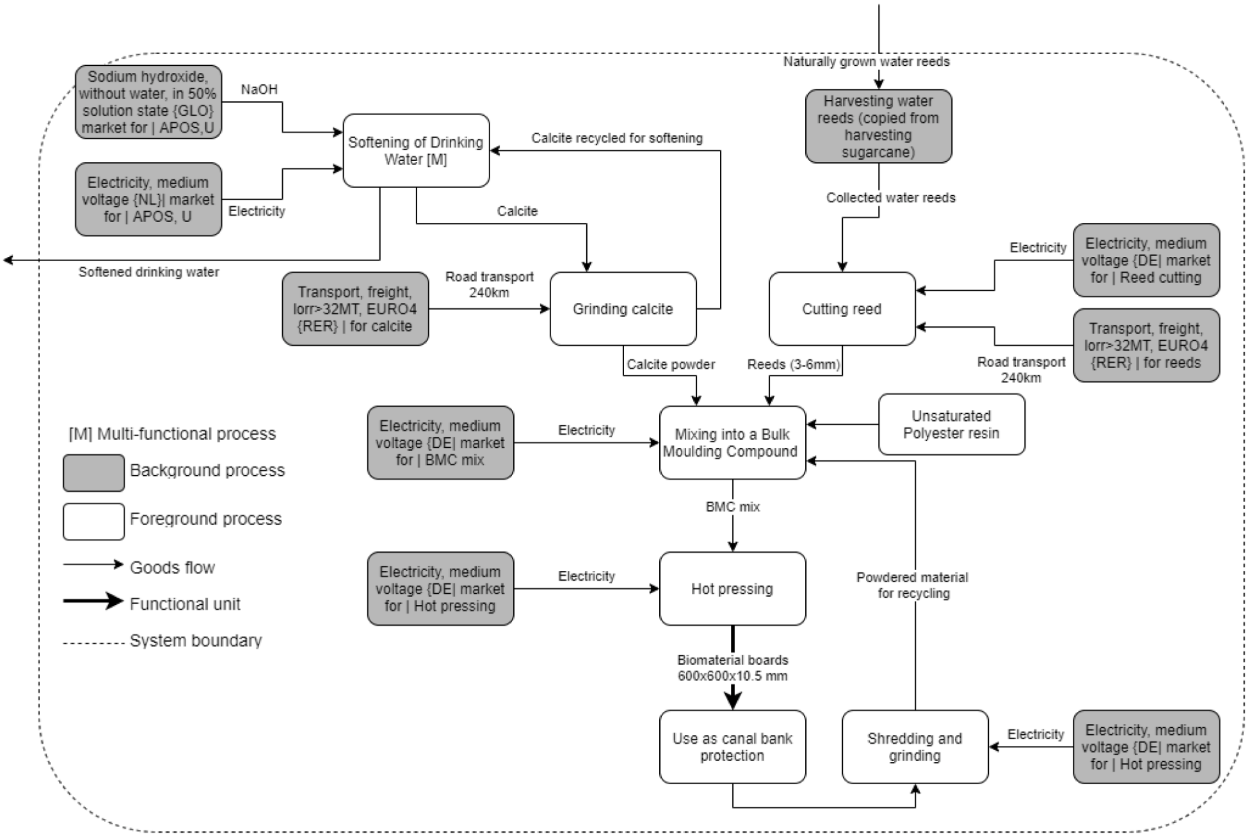

Fig. 2. Flow chart representing the system considered for a cradle to cradle LCA of NPSP bio-composite's application as canal bank protection. The system boundary begins with raw material extraction from the urban water system. It proceeds with pre-treatment of raw materials (grinding calcite and cutting reed), followed by mixing into a BMC and hot pressing. The material is used as canal bank protection. The last process is shredding and grinding the material to use as feedstock for new production.

3.4. Multi-Criteria decision analysis

Multi-criteria decision analysis (MCDA) method combine results from multiple dimensions of sustainability (Dorini et al., 2011). But before an MCDA technique is employed, the results of economic and environmental assessments must be in comparable units.

Environmental impact and economic cost are expressed in incomparable units. Normalization is used to combine disparate units into a single indicator. It helps to interpret results and to rank alternatives in case multiple materials are available for comparison. For normalization, environmental damages have been converted into external costs using shadow prices developed by De Bruyn et al. (2018). External costs assign a monetary value to LCA impact categories based on the welfare loss caused by the environmental damage (De Bruyn et al., 2018). It must be noted that external costs of only fourteen impact categories are being used that are presented in De Bruyn et al. (2018). These are listed in Supplementary Material 2.10.

A linear additive aggregation shown in Eq. (1) is used to demonstrate the most common approach to MCDA. This provides a single composite indicator describing the sustainability of the system. The goal of decision-makers will be to minimize the combined values of environmental damage (ED) and economic cost (EC) so, the lower the value of the composite indicator (CI), the higher the system sustainability.

$$CI = ED + EC \tag{1}$$

4. Case study results

4.1. Life cycle impact assessment

4.1.1. Characterization

Life cycle inventory of elementary flows is converted into environmental impacts through characterization factors in life cycle impact assessment (LCIA). In this section, we present the characterized environmental impacts. In Table 2, we only show a few selected impact categories that will be used in our discussion on some critical characteristics of LCSA. For the complete list, the reader is directed to Supplementary Material Table S1.1.

4.1.2. Normalization

Characterized results for the impact categories are of incomparable units. To compare their relative magnitudes, normalization is performed. In normalization, the impacts of a system are compared to those of a reference average, like a country or the world (Hauschild et al., 2017). For the ReCiPe 2016 method, normalization factors were introduced by the method developers in 2010, to normalize impacts using the global average per capita data. This World 2010 set of normalization factors are used here due to their compatibility with the ReCiPe method and because, global factors are the most justifiable choice from a scientific point of view (Sleeswijk et al., 2008). Table 3 presents normalized impacts for the seven highest-impact categories..For the rest, Supplementary Material S2.5 may be consulted.

Four impact categories stand out with the highest normalized magnitude. These are marine ecotoxicity (MET), freshwater ecotoxicity (FET), terrestrial ecotoxicity (TET), and human toxicity (HT) (combined impact of non-carcinogenic and carcinogenic human toxicity). FET and

Table 2
Life Cycle Impact Assessment results for the bio-composite application to canal bank protection. The table shows impact categories, their units, and values, the primary responsible substance and activity from the life cycle.

Impact category	Unit	Impact	Major elementary flow-level contribution (compartment)	Major activity-level contribution
Global warming	kg CO_2 eq.	8.77E + 01	CO_2, fossil (air)	82% Polyester resin production
Freshwater eutrophication	kg P eq.	3.62E-02	Phosphate (water)	43% Polyester resin production 36% Shredding and grinding for recycling
Terrestrial ecotoxicity	kg 1,4-DCB	2.22E + 02	Copper (air)	86% Polyester resin production
Freshwater ecotoxicity	kg 1,4-DCB	3.70E + 00	Copper (water)	78% Polyester resin production
Marine ecotoxicity	kg 1,4-DCB	4.84E + 00	Copper (water)	77% Polyester resin production
Human toxicity	kg 1,4-DCB	8.00E + 01	Chromium VI/Zinc (water)	68% Polyester resin production 19% Shredding and grinding for recycling

Table 3
The seven highest environmental impacts based on normalization to global average 2010 per capita impacts.

Impact category	Unit	Normalized impact
Marine ecotoxicity	Person.year (2010 World)	4.69E + 00
Freshwater ecotoxicity	Person.year (2010 World)	3.02E + 00
Human carcinogenic toxicity	Person.year (2010 World)	1.04E + 00
Human non-carcinogenic toxicity	Person.year (2010 World)	5.17E-01
Terrestrial ecotoxicity	Person.year (2010 World)	2.14E-01
Freshwater eutrophication	Person.year (2010 World)	5.58E-02
Global warming	Person.year (2010 World)	1.10E-02

MET are mainly caused by emission of copper into water, which accounts for about 80% of the total impact. TET is also due to copper emission but into air. For the HT impact, human carcinogenic toxicity can almost entirely be attributed to chromium VI emission into water. Non-carcinogenic toxicity is mainly due to zinc emission into water. All these emissions can be traced back to the manufacturing of polyester resin.

Freshwater eutrophication (FE) is almost entirely contributed (~98%) by phosphate emissions to water. Global warming (GW) is mainly due to fossil CO_2 emissions to air. Also, these two impacts can be traced back to polyester resin manufacturing which is the hotspot of the bio-composite life cycle.

4.1.3. *Contribution analysis*

The purpose of contribution analysis is to identify the contribution of different activities from the life cycle towards an environmental impact. In Fig. 3, we show the contribution analysis results for the category of global warming. This category has been selected due to the presence of negative emissions which will be part of our discussion later. The rest of the categories are discussed in Supplementary Material S1.6.

4.1.4. *Comparison with an alternative solution*

Assessment of hardwood as an alternative canal bank protection material was conducted to observe how LCA results can be used when multiple alternatives are compared. LCIA results for the hardwood can be found in Supplementary Material 2.11. Hardwood's global warming impact is 22.2 kg CO_2 eq. while that of the bio-composite is 87.7 kg CO_2 eq.

4.2. *Life cycle costing*

The costs associated with 6.96 standard boards are presented in Table 4. As shown, the total life cycle cost adds up to € 200.97 for 6.96 standard boards. It must be noted that discounting of future costs was not taken into account for simplicity.

Table 4
Life cycle cost of 6.96 standard bio-composite boards (600 × 600 × 10.5 mm).

Cost item	Cost (€)
Raw material (purchase and pre-treatment)	10.51
Transportation of raw material	3.42
Manufacturing (energy and material)	178.18
Installation at canal side	4.41
Recovery and recycling	4.45
Total cost	200.97

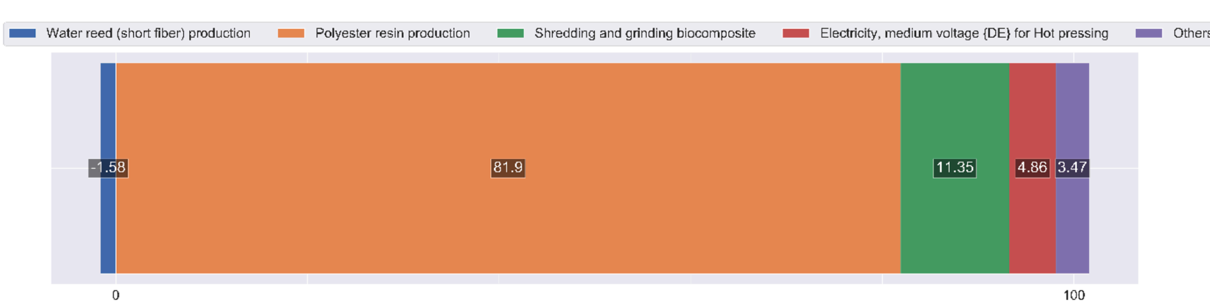

Fig. 3. Global warming impact contribution from parts of the bio-composite life cycle. All the processes with contributions less than 1% each have been clubbed under 'others' for ease of depiction on the graph.

4.3. Multi-Criteria decision analysis

Environmental impacts calculated with the LCA were converted into external costs based on shadow prices developed by De Bruyn et al. (2018). The total external cost of the life cycle is € 21.15. For the complete calculation, Table S2.12 may be consulted.

Environmental damage shadow prices and economic cost obtained from Section 4.2 are added to express sustainability in a single composite indicator as in Eq. (2):

$$CI = EC + ED = 200.97 + 21.15 = € \ 222.12 \tag{2}$$

A total aggregated score of € 222.12 is obtained, combining undiscounted direct economic and external environmental costs. So, can this be considered a sustainable solution on its own, or if there is an alternative material for canal bank protection, can it help compare the alternatives for sustainability performance? In the next section, we continue our discussion on the characteristics of LCSA in light of this case study. The order of discussion will be methodological-conceptual-ontological for ease of linking with the case study results.

5. Discussion

5.1. Methodological characteristic of LCSA

In Section 2.2.3, a methodological characteristic of LCSA was mentioned: equating a context-less eco-efficiency with environmental sustainability. Our case study demonstrates that LCA calculates environmental impacts per unit human demand (1000 mm of canal bank protection), also known as eco-efficiency. This helps to understand two factors of environmental performance. Firstly, it reveals the most significant impacts. In our case study, we find ecotoxicities as the most significant environmental impacts. Next, it reveals that polyester resin manufacturing is the most environmentally damaging process or the hotspot in the life cycle. Information about the most significant impacts and hotspots can help improve eco-efficiency. Instead of polyester resin, an alternative resin may be used with lower impacts. Eco-efficiency can also help compare the bio-composite to an alternative such as hardwood to ascertain the material with a lower environmental impact. But, eco-efficiency alone does not convey a complete picture of sustainability (Garnett, 2014; Hauschild et al., 2017; Pelletier et al., 2019). Moreover, resource recovery solutions are meant to manage resources better and avoid transgression of environmental thresholds.

Equating eco-efficiency with sustainability leads to two problems, as discussed in Section 2.2.3. Firstly, emission values alone without context can potentially underestimate the urgency of environmental damage. To contextualize a global warming potential of 87.7 kg CO_2 eq. (Table 2), current emissions and emission thresholds should be established. To demonstrate one way of doing this, we calculated the total GHG emissions from building bio-composite canal bank protection for all the canal ways in the Netherlands. The total canal bank length in the Netherlands is approximately 12,000 km (Statista, 2021), and GHG emission per metre canal bank protection using the bio-composite was 87.7 kg CO_2 eq. Taking the population of the Netherlands to be 17,167,160 (Worldometer, 2021) and the lifespan of the canal bank protection to be 25 years, the GHG emission per capita is found to be 0.00245 t CO_2 eq./capita/year. The current GHG emissions attributed to the Netherlands are 11.4 t CO_2 eq./capita/year (Centraal Bureau voor de Statistiek 2021). Although emissions from using the bio-composites as canal bank protection are minimal compared to the actual emission rates attributed to the Netherlands, the message is very different when we include the remaining budget of GHG emissions. There are several ways to calculate the emission budget of a nation. Here, we select the method described by Romanovskaya & Federici (2019) that calculates the remaining per capita GHG emission budget based on socio-economic factors (population, per-capita GDP, current net GHG emissions) and physical factors (temperature, population density). Based on their methodology, the EU member states have been assigned a budget of 22.8 Gt CO_2 eq. from 2014 through 2100 to avoid a temperature rise greater than 1.5 °C over pre-industrial levels (threshold). Thus, an equitable GHG emission budget of 0.59 t CO_2 eq./capita/year for the European population may be considered, based on the EU population of 447,706, 200 in 2020 (Eurostat, 2020). In this context, where the emission budget is already being exceeded, even a small net positive emission of 0.00245 t CO_2 eq./capita/year is only making a dire situation worse. The above calculation shows that in the context of current GHG emission levels of 11.4 t CO_2 eq./capita/year, a net positive emission of 0.00245 t CO_2 eq./capita/year may not be considered sustainable for resource recovery solutions aimed at avoiding transgression of environmental thresholds.

Secondly, even in cases with multiple alternatives, comparing their eco-efficiencies without considering environmental thresholds is insufficient. Assuming transgression of environmental thresholds will lead to calamities, then a more eco-efficient system only postpones the calamity buying us some time. As discussed above, the GHG emission budget allows for 0.59 t CO_2/capita/year while current emissions lie around 11.4 t CO_2/capita/year. Even if hardwood allows for lower emissions, as shown in Section 4.1.4, it is incorrect to call it sustainable while it contributes to the overshoot of the GHG emissions budget.

Two issues with equating a context-less eco-efficiency and sustainability were pointed out in this section. Its inability to provide context to assess sustainability leads to potential underestimation of environmental risks. With multiple alternatives, it cannot evaluate whether a more eco-efficient alternative can avoid transgression of environmental thresholds. Therefore, a context-less eco-efficiency alone is insufficient to assess sustainability. A question may be raised: Besides eco-efficiency, what else must be considered? This is covered in the next section.

5.2. Conceptual characteristic of LCSA

A pure damage assessment approach like eco-efficiency is insufficient to avoid the transgression of thresholds and ignores the possibility of actively enhancing nature. From Table 2, it is clear that the LCSA approach to environmental sustainability is one of damage reduction. For example, through an LCA, eutrophication damage of 3.62E-02 kg P eq. is calculated, and its responsible activities and substances, respectively, resin manufacturing and phosphate, are determined. This helps reduce the damage by using other resins, thus improving eco-efficiency in terms of eutrophication. However, the focus remains on damage reduction and not reciprocity.

Mutually-benefiting symbiosis in nature has a necessary element of reciprocity—for example, arbuscular mycorrhizal fungi and plants. Plants help fix atmospheric carbon for the fungi, and in return, the fungi supply nutrients to the plants (Lutzoni et al., 2018). Explicit consideration of reciprocity from industrial symbiotic systems towards nature is not found in the LCSA framework applications.

The resource recovery solution discussed in this paper has a small element of reciprocity towards nature in the form of negative CO_2 emission due to using water reeds in the bio-composite production, which absorb CO_2 from the atmosphere to grow. We will explain this reciprocity element from our case study. The carbon cycle will be used as an example which can be modelled using a five reservoir system, namely geosphere (fossil fuel reserves, sedimentary rocks, marine sediments), biosphere (living & dead biomass, soil, ecosystems), atmosphere, oceans, and anthroposphere (products, stockpiles, wastes). We use this model because it covers all the relevant reservoirs in a time frame of years to centuries (Ajani et al., 2013).

Any anthropogenic emission of CO_2 is a transfer of carbon from either biosphere or geosphere to the atmosphere (Ajani et al., 2013). Negative emission, crucially, helps to transfer excess CO_2 out of the atmosphere. We suggest it is, in fact, essential to sustainability since it is in line with our understanding of the transgressed planetary boundaries of some environmental categories. The planetary boundary zone of climate change is commonly accepted as 350–450 ppm atmospheric concentration of CO_2 (Steffen et al., 2015). It is also known that the lower limit of this zone has been crossed, as atmospheric CO_2 concentration lies around 420 ppm as of April 2021 (Scripps institution of oceanography, 2021). To continue designing 'sustainable solutions' without explicit consideration for how it affects the carbon cycle is inconsistent with our appreciation of climate change thresholds.

While emission reduction is crucial, it is insufficient, especially for impact categories that have crossed or neared environmental thresholds. To differentiate from emission reduction, we introduce the sub-concept of 'Reciprocity' into the broader concept of industrial symbiosis. With regards to the carbon cycle, it refers to the anthropogenic redistribution of carbon stocks out of the atmosphere into other reservoirs, intending to restore the carbon cycle such that the climate system is not disrupted irreversibly. But, simply modelling this redistribution as negative emission is not enough. There are two reasons for this.

Firstly, clubbing eco-efficiency and reciprocity hides necessary distinctions for proper reciprocity considerations, such as details of the stocks and their lifetime. In symbiotic solutions such as our case study, a two-way flow of CO_2 is present, i.e., CO_2 is emitted as a result of fossil-based electricity but also absorbed as a result of bio-based raw materials like water reeds. Whenever a two-way flow is a possibility, details of the reservoirs in terms of stability, residence time, etc., must be kept in view since carbon reservoirs are not fungible (Ajani et al., 2013). But, eco-efficiency does not account for these details. To explain further, biogenic carbon absorption is modelled chiefly as an offset to fossil fuel emissions. The characteristics of a carbon reservoir decide its effects on atmospheric carbon concentrations and, thus, climate change (Ajani et al., 2013). The residence time of carbon stocks in the geosphere runs into millions of years, while that of the stock in the biosphere is only about 23 years (Schlesinger and Bernhardt, 2020). Thus, fossil carbon emission speeds up the carbon cycle a lot more than biosphere emission. Hence, we cannot directly compare the negative emission from using reeds and positive emissions from fossil fuel burning as in Fig. 3.

Secondly, given that reciprocity is essential, considering the carbon cycle in terms of stocks instead of merely negative emissions helps design sustainable systems that proactively mitigate climate change. For example, in our case study, it can provide answers to questions like what quantity of reeds to use and how long the product needs to remain in the anthroposphere to offset its fossil fuel emissions.

In this section, we discussed that it is not enough to minimize damages caused to nature but that it is necessary to restore its capacity to support human life actively. This is especially true for environmental damages that have crossed thresholds. The LCSA framework assumes that a sustainable solution is one that minimizes environmental impacts per unit demand. This assumption should not be the basis of sustainability assessments for resource recovery solutions that may potentially have positive impacts on nature. There is a need for including positive contributions of humans or reciprocity towards nature in the related assessment and methodologies to include it may be developed.

5.3. Ontological characteristic of LCSA

From the environmental and economic assessment, two factors of sustainability become evident: Environmental impacts of fulfilling a unit human demand and its life cycle cost. Two indicators, namely environmental damage and economic cost, were normalized and aggregated in Section 4.3. Linear additive aggregation was used as it is the most common kind. But, such an aggregation assumes that a lower life cycle cost can make up for higher environmental damage. As an example, LCC of hardwood was obtained through personal correspondence with Waternet as € 73.33. A simple LCA was conducted for hardwood application for canal bank protection, and results were converted into an external cost value of € 90.15. Details of this calculation can be found in Supplementary Material 2.12. A linear additive aggregation provides a composite indicator value of € 163.48, a lower and, thus, more sustainable value compared to the bio-composite (€ 222.12). From this example, it is clear how the higher environmental damage of hardwood application to canal banks can be compensated by its lower life cycle cost. In this manner, compensation between the pillars of sustainability is commonly operationalized in literature. The compensation between criteria stems from a weak sustainability notion.

In Section 2.2.2, we gave three arguments why weak sustainability is unfounded. Firstly, our economic activities rely on nature and its processes. This is obvious in the case of resources obtained from nature. For example, coal for electricity production is obtained from natural reserves. Also, we depend on the natural processes that support their growth for raw materials such as water reeds. For instance, the natural cycles of carbon and nutrients like phosphorus and nitrogen are indispensable for the reeds to grow. Given that continued manufacturing of the bio-composite relies on the natural processes, health of the natural process and continued economic activity are not substitutable. It is not sensible to allow any economic welfare obtained from manufacturing to compensate for the irreversible disruption of natural processes.

Secondly, we had pointed out that there is a fundamental difference between natural capital and economic capital. The former is characterized by irreversibility and thresholds, whereas built capital can be rebuilt and is never irreversibly lost as long as natural capital is available. For example, the major environmental damage of our case study is freshwater eutrophication, caused primarily by phosphate emissions. Eutrophication makes water non-potable, causes human toxicity due to cyanobacterial bloom, causes fish kills, and is extremely expensive to mitigate (Carpenter and Bennett, 2011; Smith et al., 2006). The effects of eutrophication are subject to irreversibility in case a threshold value of emission is transgressed (Carpenter and Lathrop, 2008). Consequently, eutrophication has the potential to change the physio-chemical regime of ecosystems permanently. Whether the new ecosystem regime would allow for our survival, let alone, continued manufacturing industry, is disputable.

Lastly, we had mentioned how natural capital fulfils multiple functions that cannot be replaced by anthropocentric built capital. As a result of irreversible eutrophication, fish kill is expected. Fish have a role in maintaining a healthy food-web, their waste is a source of nutrition for aquatic plants, and fishing is a source of livelihood for many. Loss of all these functions cannot be practically compensated by any single built capital that replaces it, let alone a bio-composite meant for canal bank protection.

Due to the three reasons discussed above, economic welfare and environmental damage are not entirely substitutable, and hence, the weak sustainability notion is found wanting. What notion of sustainability is being followed has a great impact on understanding and assessing sustainability (Huang, 2018). Since the notion of weak sustainability is unfounded, the compensation between economic welfare and environmental damage is unjustified. Daly & Goodland (1996) presented two alternatives to weak sustainability: absurdly strong and strong. Absurdly-strong sustainability does not allow any trade-off between environmental damage and economic development. Decision-making, after all, is about trade-offs, so they may not be totally avoidable. Strong sustainability allows for substitutability at a certain level. How to define this level of substitutability requires further research before LCSA can be applied to resource recovery solutions.

6. Future research directions

From our review of LCSA applications to water sector resource recovery solutions and evaluation of the framework through our own application on a real-life case study, the strengths of LCSA are evident. The framework allows for an assessment over the entire life cycle of a resource recovery solutions, thus preventing shifting of burden between different life cycle stages. Through its inclusion of a wide range of impact categories, it also avoids burden shifting between different kinds of environmental damages. However, certain characteristics of LCSA may be modified to make it better suited to resource recovery solutions.

In this regard, we propose three future research directions below.

Firstly, two issues were pointed out in equating eco-efficiency with sustainability. When assessing a resource recovery solution, emission intensity alone does not convey enough information to judge its sustainability. CO_2 emission from using bio-composite canal banks was used to demonstrate this. When this emission value is placed in the context of current emission rates and the remaining emission budget of the Netherlands, an overshoot of the emission budget and thus its unsustainability becomes evident. Also, when multiple alternatives are present, eco-efficiency helps ascertain the alternative with a lower impact but is insufficient to judge their absolute sustainability. In this case, calling one of the alternatives more sustainable may be misleading. Therefore, following research directions are suggested.

Characterized LCA results of water sector resource recovery solutions, as shown in Table 3, can benefit if placed in a context. Context can be provided by including past emissions and environmental thresholds. Thresholds can be based on the planetary boundaries from Rockström et al. (2009) for global environmental issues such as climate change. Methods need to be developed to scale down planetary boundaries logically before applying them to resource recovery solutions in the water sector. We demonstrated a method proposed by Romanovskaya & Federici (2019) to scale down carbon emission budgets for the EU population. In this manner, the assessment can estimate the impact of resource recovery and reuse on the absolute sustainability of carbon emissions. Ryberg et al. (2018) describe another method to link LCA results to planetary boundaries by assigning a 'safe operating space' to laundry washing. Principles for calculating such a safe operating space for resources involved in the water sector need defining. This helps link characterized results of LCA to absolute limits and ascertain if a resource recovery solution allows the water sector to actually avoid transgression of environmental thresholds and for how long.

Secondly, it was shown that improved eco-efficiency proves insufficient in some instances. CO_2 emission was shown as an example for which mere emission reduction may no longer be sufficient to avoid transgression of planetary limits. It was discussed how relying on negative emissions neglects crucial details about the carbon reservoirs. Furthermore, it was mentioned that it might be possible to proactively design sustainable solutions to mitigate climate change if a dynamic carbon cycle model is included in the assessment methodology. The authors suggest incorporating such a dynamic flow model and a new set of indicators to quantify 'reciprocity' of resource recovery solutions. Some research ideas on this topic are presented below.

Since resource recovery solutions can provide services towards nature, a framework can benefit from having a methodology that assesses these reciprocities towards nature in detail. For this, stock and flow models of the major biogeochemical cycles, such as carbon, nitrogen, etc., can be incorporated into sustainability assessment frameworks. The effect of a resource recovery solution on the stocks and flows of these cycles can be estimated using dynamic models. Therefore, future research must be conducted to identify/develop models that can be used for this purpose. Le Noë et al. (2017) used a nutrient flow model called Generalized Representation of Agro-Food System (GRAFS) to assess environmental impacts of different kinds of French agricultural systems. Along similar lines, flow models for nitrogen, phosphorus, water, carbon, etc., can be instrumental in resource recovery and reuse assessment. For instance, a carbon flow model could be used for dynamically assessing the impact of a resource recovery solution on the carbon cycle. In our case study, it could provide details such as duration, quantity, and quality (biogenic or fossil) of carbon sequestration, which are critical distinctions (Ajani et al., 2013). Knowing these details about the different stocks and fluxes can make it easier to conceptualize proactive solutions for redistributing carbon out of the atmosphere. Research is also needed to develop indicators pertaining to these biogeochemical models that will help quantify improvements that a resource recovery process has on these biogeochemical cycles. This will also help overcome a major limitation of the eco-efficiency approach, being limited to calculating and reducing negative impacts (Nika et al., 2020).

Thirdly, consequences of linear additive aggregation were shown to result in unjustifiable compensation between economic and environmental capital. Limitations of allowing complete compensation between the environmental and economic criteria were discussed. Three issues were pointed out with comparing natural and built capital. Firstly, the dependence of the economic system on the natural environment, secondly the possibility of irreversible damages to natural capital, and thirdly, the multi-functionality of natural capital that sets it apart from built capital.

Any assessment framework must avoid making trade-offs between criteria that are not substitutable. Then again, it is impractical to conceive of any economic activity without some environmental modification. Therefore, a 'discriminate trade-off' methodology becomes necessary. There are methods to aggregate and rank alternatives in ways that allow for lower degrees of compensation or even to avoid it altogether (Polatidis et al., 2006), but discussions on the extent of compensation used and the rationale behind it is missing in most studies. Generally, the subject of compensation in MCDA has not received much attention (Guitouni and Martel, 1998), and studies aggregating in a non-compensatory fashion are few (Burgass et al., 2017). Criteria must be defined to segregate compensable and non-compensable environmental damages. The critical natural capital framework may be used as a starting point for this. The use and further development of non-compensatory aggregation methods in sustainability assessment, such as PROMETHEE, ELECTRE, etc., also need to be researched.

7. Conclusions

Our aim with this paper was to analyse the strengths and weaknesses of the existing LCSA framework, in the case that it is used to assess the sustainability of water sector resource recovery solutions and to suggest necessary modifications, if required. Three characteristics of LCSA were identified that could be modified to serve resource recovery solutions better. These were categorized as conceptual, ontological, and methodological. These were evaluated using a real-life case study where canal bank protection elements made using a new bio-composite material (obtained via resource recovery) were compared to the elements made of conventional hardwood. The three LCSA characteristics can be modified and some corresponding future research directions are as follows:

1 Conceptually, the existing LCSA framework remains focussed on assessing negative impacts of humans on nature. We demonstrated how positive impacts of resource recovery solutions are expressed in terms of negative emission, which hides crucial details about the dynamics of natural biogeochemical cycles. This damage-based assessment framework can benefit from including the concept of reciprocity, i.e., positive contribution of humans towards nature. New methodologies and indicators to assess reciprocal benefit from humans to nature in detail need to be developed. This 'reciprocity' assessment can be achieved through dynamic flow models of elements such as carbon, nitrogen, etc., and developing indicators to measure restoration of stocks and flows of biogeochemical cycles.

2 A methodological type limitation of the existing LCSA framework is the use of LCA without using thresholds and current emissions/resource stocks. We demonstrated how expressing sustainability in terms of emissions or resource consumption alone can potentially underestimate environmental issues and possibly neglect one goal of resource recovery solutions: avoiding environmental threshold transgressions. Eco-efficiency assessment should be combined with assigning a 'safe operating space' to resource recovery solutions. This requires developing principles for scaling down the remaining emission/resource use budgets available for a particular resource recovery solution. Including information on past emissions and

environmental thresholds in eco-efficiency measurements is also recommended.

3 Finally, from the ontological point of view, a characteristic of the existing LCSA framework is that it treats the economic and natural capitals as completely substitutable. This is unjustifiable due to the reliance of economic capital upon nature, the presence of thresholds related to natural capital, and the multi-functionality of natural capital that cannot be compensated for by built capital as discussed in the case study. Methods to limit the extent of this substitutability need to be developed for resource recovery solutions. Concepts of critical natural capital and use of non-compensatory MCDA methods may be helpful in this regard.

Future work will focus on developing a new assessment framework for water sector resource recovery solutions that addresses the characteristics of LCSA discussed above.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2021.106151.

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