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Circularity & Efficiency Assessment of Resource Recovery Solutions for the Water Treatment Sector

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Circularity & Efficiency Assessment of Resource Recovery Solutions for the Water Treatment Sector

Dissertation

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Contents

List of Figures

List of Tables

List of Acronyms

AHP Analytical hierarchy process

AS Activated sludge

BA Biomass nutrient assimilation

BW Black water

COD Chemical oxygen demand

DAF Dissolved air floatation

DSS Decentralized source separation

DWR Drainage water recycling

EC Electro-coagulation

ET Evapotranspiration

FR Freshwater restoration

FW Freshwater

GDP Gross domestic product

GHG Greenhouse gases

GW Grey water

GWF Grey water footprint

IA Intermittent aeration

IE Irrigation efficiency

LCA Life cycle assessment

LCC Life cycle cost

LCI Life cycle inventory

LCIA Life cycle impact assessment

LCSA Life cycle sustainability assessment

MAUT Multi-attribute utility theory

MCDA Multi-criteria decision analysis

MCI Material circularity indicator

MFA Mass flow analysis

NRE Nutrient recovery efficiency

NUE Nutrient uptake efficiency

OM Organic matter

OSA Oxic settling aeration

PPP Purchasing power parity

S-LCA Social-Life cycle assessment

SOC Soil organic carbon

SOM Soil organic matter

SRT Solid retention time

SS Soil organic matter sequestration

STAN Substance flow analyser

TD Tile drainage

TN Total nitrogen

TP Total phosphorus

TW Treated wastewater

UASB-ST Upflow Anaerobic Sludge Blanket-Septic tank

UF Ultrafiltration

VS Volatile solids

WFE Water-food-energy

WPL Water pollution level

WWTP Wastewater treatment plant

Summary

Water treatment and drinking water production plants are transitioning to resource recovery facilities to prevent pollution and resource depletion. Resources such as treated wastewater, nutrients, organic matter, and calcite may be recovered from wastewater and drinking water production plants. Assessment methods are needed for the decision-makers to evaluate and compare the performances of different resource recovery solutions. The aim of this thesis is to develop methods to assess the circularity, efficiency, and nature reciprocity of these resource recovery-based solutions.

Chapter 2 answers the research question: *What are the strengths and weaknesses of life cycle sustainability assessment in the context of resource recovery solutions in the water sector?*

The life cycle sustainability assessment (LCSA) framework comprises of the life cycle assessment (LCA), life cycle costing (LCC), and social life cycle assessment (S-LCA) methods to assess the performance of a resource recovery solution along the environmental, economic, and social sustainability dimensions. This framework is useful to prevent burden-shifting and assists in a comprehensive assessment. Nevertheless, some limitations of the existing framework are identified with respect to three crucial characteristics of the resource recovery solutions in the water sector. The limitations are discussed as a starting point to develop the new methods for the circularity, efficiency, and nature reciprocity assessment.

The three crucial characteristics of the resource recovery solutions in the water sector are as follows: (i) they have the potential to actively benefit the natural environment, (ii) they depend upon natural resources and processes, and (iii) their goal is to avoid the transgression of environmental thresholds. Next, the limitations of the LCSA framework in taking account of these characteristics are presented under three categories: conceptual, ontological, and methodological.

Conceptually speaking, LCA is a method to assess environmental damages. Therefore, the use of LCA ignores the possibility of the resource recovery solutions to provide benefits to the natural environment. Ontologically, the LCSA framework considers the natural environment as something that is substitutable with economic growth. This viewpoint is inconsistent with the fact that future economic development is completely dependent on the preservation of natural capital. Methodologically, the LCA indicators reported in literature often lack the necessary context of current emission levels and environmental thresholds for these emissions.

Chapter 2 concludes with a call to develop methods that can estimate the nature benefits from resource recovery, multi-criteria decision analysis techniques that limit the compensation between the economic and the environmental criteria, and to include the contextual information in LCA analyses in future studies.

In chapter 3, the following research question is answered: *How can a method be developed to accurately assess the circularity of the biogeochemical resources present in the water treatment plants?*

A novel circularity assessment method is developed for the water treatment sector. This sector deals with a mix of technical and biogeochemical cycle resources. The technical cycle resources are generally abiotic, non-renewable, and synthetic (e.g., metals and plastics) and have the potential to remain circulating in the production system (i.e., industrial manufacturing, recovery, and reuse), without being disposed in landfills or used as fuel for energy generation. Contrarily, the biogeochemical cycle resources (e.g., water and nitrogen) pass alternatively between non-living forms and as parts of living organisms. Since most of the current methods were developed for the technical cycle resources, they tend to ignore the complexities of the biogeochemical cycles.

The need for a separate circularity assessment method for the biogeochemical cycle resources is motivated by three reasons. Firstly, differentiating between linear and circular flows for biogeochemical resources is more complex than for the technical cycle resources. Secondly, humans use the biogeochemical resources for their needs, but these are also essential for the natural environment. Maximising their use for human purposes can potentially make them scarce for the natural environment. Thirdly, the environmental losses of biogeochemical resources can be significant and need to be considered for an accurate circularity assessment. To address these issues, a new method is developed by modifying the existing Material Circularity Indicator (MCI) method through redefining restorative, regenerative, and linear flows for the biogeochemical resources.

The modified MCI method is applied to a case-study using treated wastewater for fertigation (applying fertilizers with irrigation water) in Corleone, Italy. The assessment revealed that the water and nitrogen circularities improve if treated wastewater is used for fertigation instead of freshwater with industrial fertilizers. It was also found that a low-frequency application (every ten days) is favourable in terms of higher water circularity but the nitrogen circularity is favoured by a higher application frequency (every three days) for the particular combination of crop and farming conditions.

The chapter concludes by emphasizing the need for more accurate and accessible resource flow models to be coupled to the newly developed circularity assessment method.

Chapter 4 deals with the following research question: *What are the potential nature benefits from the application of the recovered resources and how can they be maximized?*

The concept of nature reciprocity is introduced along with an innovative method to for its assessment. Nature reciprocity is defined here as the re-balancing of the biogeochemical resource stocks by directing the resource from one stock to another that could benefit from such a transfer, e.g., soil sequestering of the excess carbon in the atmosphere. Nature reciprocity is necessary in cases where the resource imbalance has crossed the planetary thresholds, such as the excess reactive nitrogen build-up in the terrestrial and aquatic environments. The assessment method involves three new indicators: freshwater restoration (FR), biomass assimilation of nutrients (BMA) and soil organic matter sequestration (SS).

This method accounts for the quantity of the recovered resources as well as particular characteristics that ensure the maximum nature benefit. For the FR assessment, the treated wastewater quality is included in terms of the water pollution level. The lower the pollution level (i.e., the better the treated wastewater quality), the higher would be the FR. For the BMA, the nutrient recovery efficiency (NRE) is combined with the nutrient uptake efficiency (NUE). The latter characterizes how easily the recovered nutrients can be taken up by crops. The higher the crop uptake the higher would be the BMA. Similarly, for SS, the mass of the organic product applied to the soil is combined with its volatile solids composition to characterize its stability. A higher stability implies a higher SS.

This method is applied to a theoretical case-study with a WWTP located in Wilp, the Netherlands. This WWTP is designed with the focus on resource recovery and relies predominantly on physio-chemical processes, including electro-coagulation, nanofiltration, and ion-exchange instead of the more commonly used activated sludge process. The only biological process used at Wilp is the anaerobic digestion of the excess sludge.

The assessment of Wilp revealed that the WWTP's performance is of a sufficient level ensuring that most of the influent wastewater (92%) is converted into restored freshwater discharged into River Ijssel. Further, a higher BMA is achieved by struvite recovery compared to vivianite recovery. This is due to a higher NRE using the struvite precipitation (80%) than the viviante recovery (64%). Furthermore, Wilp can also contribute to the soil sequestration of 7.3×10^5 kg/y organic matter by using the anaerobically digested sludge. This translates to about 4.4 kg/y/capita carbon (assuming 60% carbon content). This could have a positive effect on the local soil quality by restoring its organic matter content. The results obtained demonstrate that the proposed method can be used to recognize and assess the potentially positive role that humans can play in the natural environment.

In Chapter 5, the following question is answered: *How can the conventional centralized and the decentralized source separation treatment* *approaches be compared from a water-food-energy perspective in an integrated and holistic way?*

A new water-food-energy (WFE) nexus framework is presented to compare a conventional centralized WWTP and a decentralized source separation (DSS) one. It contains novel indicators to capture the benefits to the water treatment and the food production sectors resulting from resource recovery solutions.

The framework is applied to two case-studies serving 12000 p.e.: a conventional centralized WWTP located in Corleone, Italy, and a DSS one in Helsingborg, Sweden. Corleone uses a centralized activated sludge (AS) treatment with intermittent-aeration, an oxic-settling anaerobic tank, an ultrafiltration unit to produce irrigation water for farms located 2 km away from the WWTP. The treated wastewater will be partially discharged into a stream and partially used for agricultural irrigation. Helsingborg separates black and grey water using vacuum toilets. The grey water is transported using a low-pressure sewer and treated in an AS reactor. The treated grey water is discharged into the ocean but plans to reuse it for irrigating farms are being discussed. A distance of 0.1 km between the WWTP and the farms has been assumed here. The black water is anaerobically digested with the excess sludge from the AS unit. The anaerobic digestate is used to manufacture soil and compost for agriculture. The two case studies differ in pollutant concentrations and the proportions of dissolved and particle-bound pollutants which are crucial factors for an accurate mass balance.

Comparing the water treatment performance, Corleone is better than Helsingborg on most of the efficiency and circularity indicators except for nutrient circularity. This is mainly due to the treatment/recovery processes in Helsingborg being more resource-intensive. Helsingborg performed better than Corleone on most of the efficiency and circularity indicators related to food production. This is mainly because of the arid climate and the longer distance between farms and the WWTP in Corleone. Further, Helsingborg was better than Corleone on all the nature reciprocity indicators. Corleone had a negative freshwater restoration value of -8.1×10^7 m^3/y implying that the pollutants in the effluent require, for their dilution, a higher water flow rate than that of the stream. Finally, Helsingborg achieved a score of 212% for energy self-sufficiency while Corleone's energy self-sufficiency was 0%.

The above demonstrates that the newly developed framework helps to perform a multi-dimensional comparison between the two approaches to water treatment, accounting for the relevant climate and agricultural conditions. The use of indicators relevant to the water treatment and the food production sectors ensures easier communication and could contribute to better coordination in the future.

This PhD thesis has several scientific contributions. Firstly, the new circularity assessment method that is capable of simultaneous assessment of both technical and biogeochemical cycles resources has been developed by modifying the existing MCI method. Secondly, the nature reciprocity method is proposed. It was demonstrated that this method can reveal novel resource recovery and application pathways resulting in different, improved decision outcomes. Thirdly, a new WFE framework has been developed that is capable of a holistic and integrated comparison of conventional centralized and decentralized source separation approaches to wastewater treatment and resource recovery. Finally, various case-studies presented in this thesis contain useful data related to TW reuse, wastewater treatment approaches, and DSS WWTPs which may be used for future studies.

Regarding societal contributions, the implications of this research for the decision-makers are also presented in this thesis. The methods presented here will enable the decision-makers to assess and compare various resource recovery and application options in an improved way. These methods are particularly suited for the planning phase and can act as aids to the discussions between different stakeholders. The circularity method will encourage the decision-makers to consider the best way to return a resource to the natural environment. The nature reciprocity method will enable the decision-makers to evaluate the potentially positive impacts of a resource recovery solution on the natural environment. This aspect may be used alongside the negative impacts assessed using an LCA and this combination can lead to better decision outcomes than if only the negative impacts were considered. Lastly, the WFE framework will provide a practical and holistic way to evaluate and compare the conventional WWTPs with the decentralized source separated ones. Additionally, it will allow them to consider the local relevant conditions including distances between farms and WWTPs, agricultural land use per-capita, etc. Its application is likely to help in better inter-sectoral communication and coordination.

Based on this work, some directions for future research can be suggested. In research involving LCSA, it is crucial to include the current emissions/resource stocks values with some environmental thresholds. Further, LCSA studies should be using non-compensatory aggregation methods for the environmental and the economic criteria and the outcomes of non-compensatory and compensatory methods should be compared. For future circularity assessment studies, it is recommended to develop more accessible resource flow models that are sector-specific and capable of accounting for the locally relevant factors. Also, pot experiments are recommended to create a database containing a wide variety of agricultural conditions and the NUE values for the various nutrient products. Lastly, for an accurate economic efficiency assessment, more data and indicators are needed related to the quality and market prices of the recovered products.

To conclude, the planning of resource recovery solutions in the water treatment sector can substantially benefit from the methods developed in this research. These may be used along with the LCSA methods after the suggested improvements. These methods are fairly comprehensive, reproducible, and take into consideration, the locally relevant factors.

Samenvatting

Waterzuiveringsinstallaties en drinkwaterproductie-installaties worden omgevormd tot faciliteiten voor het terugwinnen van hulpbronnen om verontreiniging en uitputting van hulpbronnen te voorkomen. Hulpbronnen, zoals gezuiverd afvalwater, voedingsstoffen, organisch materiaal en calciet, kunnen daaruit worden teruggewonnen. Er zijn beoordelingsmethoden nodig voor beleidsmakers om de prestaties van verschillende resource recovery oplossingen te evalueren en te vergelijken. Het doel van dit proefschrift is het ontwikkelen van methoden om de circulariteit, efficiëntie en wederkerigheid met de natuur van resource recovery oplossingen in de waterzuiveringssector te beoordelen.

Hoofdstuk 2 beantwoordt de onderzoeksvraag: *Wat zijn de sterke en zwakke punten van Life Cycle Sustainability Assessment in de context van het terugwinnen van hulpbronnen in de watersector?*

Het Life Cycle Sustainability Assessment (LCSA) kader bestaat uit de Life Cycle Assessment (LCA), Life Cycle Costing (LCC) en Social Life Cycle Assessment (S-LCA) methoden om de prestaties van een resource recovery oplossing te beoordelen op het gebied van milieu-, economische en sociale duurzaamheid. Dit kader is nuttig om lastenverschuiving te voorkomen en helpt bij een uitgebreide beoordeling. Niettemin worden enkele beperkingen van het kader geïdentificeerd met betrekking tot drie cruciale kenmerken van resource recovery in de watersector. De beperkingen worden besproken als uitgangspunt voor het ontwikkelen van nieuwe methoden voor de beoordeling van circulariteit, efficiëntie en natuurlijke reciprociteit.

De drie cruciale kenmerken van resource recovery oplossingen in de watersector zijn als volgt: (i) ze hebben het potentieel om actief voordeel te bieden aan het natuurlijke milieu, (ii) ze zijn afhankelijk van natuurlijke hulpbronnen en processen, en (iii) hun doel is om het overschrijden van milieugrenzen te voorkomen. Vervolgens worden de beperkingen van het LCSA-kader bij het in aanmerking nemen van deze kenmerken gepresenteerd in drie categorieën: conceptueel, ontologisch en methodologisch. Conceptueel gezien is LCA een methode om milieuschade te beoordelen.

Conceptueel gezien is LCA een methode om milieuschade te beoordelen. Daarom negeert het gebruik van LCA de mogelijkheid dat deze oplossingen voordelen voor het natuurlijke milieu kunnen opleveren. Ontologisch gezien beschouwt het LCSA-kader het natuurlijke milieu als iets dat volledig vervangbaar is door economische groei. Dit standpunt is inconsistent met het feit dat toekomstige economische ontwikkeling volledig afhankelijk is van het behoud van natuurlijk kapitaal. Methodologisch gezien missen de in de literatuur gerapporteerde LCA-indicatoren vaak de noodzakelijke context van de huidige emissieniveaus en milieugrenzen voor deze emissies.

Hoofdstuk 2 eindigt met een oproep om methoden te ontwikkelen die de natuurvoordelen van resource recoveru kunnen schatten, multicriteria besluitvormingstechnieken die de compensatie tussen de economische en de milieudoelstellingen beperken of elimineren, en om de contextuele informatie in LCA-resultaten op te nemen in toekomstige studies.

In hoofdstuk 3 wordt de volgende onderzoeksvraag beantwoord: *Hoe kan een methode worden ontwikkeld om de circulariteit van de biogeochemische hulpbronnen in huishoudelijke afvalwaterzuiveringsinstallaties nauwkeurig te beoordelen?*

Een nieuwe methode voor circulariteitsbeoordeling wordt ontwikkeld voor de waterzuiveringssector. Deze sector heeft te maken met een mix van tech- nische en biogeochemische cyclusbronnen. De technische cyclusbronnen zijn over het algemeen abiotisch, niet-hernieuwbaar en synthetisch, en hebben het potentieel om te blijven circuleren in het productiesysteem (bijv. industriële productie, terugwinning en hergebruik), zonder dat ze worden gestort of als brandstof worden gebruikt voor energieopwekking (bijv. metalen en kunststoffen). Daarentegen bewegen de biogeochemische cyclusbronnen (bijv. water en stikstof) afwisselend tussen niet-levende vormen en levende organismen, als onderdeel daarvan. Omdat de meeste huidige methoden zijn ontwikkeld voor technische cyclusbronnen, negeren ze de complexiteiten van de biogeochemische cycli.

De noodzaak voor een aparte methode voor circulariteitsbeoordeling van biogeochemische kringloopbronnen heeft drie redenen: differentiatie tussen lineaire en circulaire stromen is complexer dan voor technische kringloopbronnen, biogeochemische bronnen zijn zowel essentieel voor menselijke behoeften als voor het natuurlijke milieu, en milieuverliezen van deze bronnen kunnen significant zijn. Een nieuwe methode, gebaseerd op een aangepaste Material Circularity Indicator (MCI), is ontwikkeld door herstellende, regeneratieve en lineaire stromen voor biogeochemische bronnen opnieuw te definiëren.

Deze aangepaste MCI-methode is toegepast op een casestudy waarbij behandeld afvalwater werd gebruikt voor fertigatie in Corleone, Italië. De beoordeling toonde aan dat de circulariteit van water en stikstof verbetert bij gebruik van behandeld afvalwater in plaats van zoetwater met industriële meststoffen. Lage toepassingsfrequentie (elke tien dagen) bevordert watercirculariteit, terwijl hogere frequentie (elke drie dagen) stikstofcirculariteit verbetert voor deze specifieke combinatie van gewas en en landbouwomstandigheden. Het hoofdstuk sluit af met de noodzaak van meer nauw- keurige en toegankelijke modellen voor resource flows te benadrukken die gekoppeld moeten worden aan de nieuwe circulariteitsbeoordelingsmethode.

Hoofdstuk 4 behandelt de volgende onderzoeksvraag: *Wat zijn de potentiële natuurvoordelen van de toepassing van de teruggewonnen hulpbronnen en hoe kunnen deze gemaximaliseerd worden?*

Het concept van natuurreciprociteit wordt geïntroduceerd, samen met een nieuwe methode voor de beoordeling ervan. Natuurreciprociteit wordt hier gedefinieerd als het herbalanceren van de biogeochemische hulpbronnen door de hulpbron van de ene voorraad naar een andere te leiden die kan profiteren van een dergelijke overdracht, bijvoorbeeld bodemsequestratie van het overtollige koolstof in de atmosfeer. Natuurreciprociteit is noodzakelijk in gevallen waarin de hulpbronnenba- lans de planetaire grenzen heeft overschreden, zoals de opbouw van overtol- lig reactief stikstof in de terrestrische en aquatische omgevingen. Natuur- reciprociteit wordt beoordeeld met behulp van drie indicatoren, namelijk zoetwaterherstel (FR), biomassa-assimilatie van voedingsstoffen (BMA) en vastlegging van organische stof in de bodem (SS).

De indicatoren houden rekening met de hoeveelheid teruggewonnen hulpbronnen en specifieke kenmerken die zorgen voor het maximale natuurvoordeel. Voor de FR-beoordeling wordt de kwaliteit van het gezuiverde afvalwater, in termen van de water pollution level, gebruikt als een factor. Hoe lager de pollution level (d.w.z. hoe beter de kwaliteit van het gezuiverde afvalwater), hoe hoger de FR zou zijn. Voor de BMA wordt de nutrient recovery efficiency (NRE) gecombineerd met de nutrient uptake efficiency (NUE). De laatstgenoemde karakteriseert hoe gemakkelijk de teruggewonnen voedingsstoffen door de gewassen kunnen worden opgenomen. Hoe hoger de gewasopname, hoe hoger de BMA zou zijn. Evenzo wordt voor SS de massa van het organische product dat op de bodem wordt aangebracht, gecombineerd met de samenstelling van volatile solids om de stabiliteit ervan te karakteriseren. Hoe hoger de stabiliteit van het organische product, hoe hoger de SS zou zijn.

Deze methode is gebruikt in een theoretische casestudy met een rioolwaterzui- veringsinstallatie in Wilp, Nederland. Deze RWZI is ontworpen met de fo- cus op terugwinning van hulpbronnen en maakt voornamelijk gebruik van fysisch-chemische processen, waaronder elektrocoagulatie, nanofiltratie en ionenuitwisseling, in plaats van het meer gebruikelijke actief-slibproces. Het enige biologische proces dat in Wilp wordt gebruikt, is de anaerobe vergisting van het overtollige slib. De beoordeling van Wilp toonde aan dat de prestaties van de nieuwe RWZI van een voldoende niveau zijn, waardoor het meeste van het binnen- komende afvalwater (92%) wordt omgezet in hersteld zoetwater dat wordt geloosd in de rivier de IJssel. Verder wordt een hogere BMA bereikt door struviet terugwinning in vergelijking met vivianiet terugwinning. Dit komt door een hogere NRE bij het neerslaan van struviet (80%) dan bij de vivia- niet terugwinning (64%). Bovendien kan Wilp ook bijdragen aan de bodem- vastlegging van $7{,}34\times10^5$ kg/j organische stof door het gebruik van het anaeroob vergiste slib. Dit komt neer op ongeveer 4,4 kg/j/capita koolstof (uitgaande van een koolstofgehalte van 60%). Dit zou een positief effect kunnen hebben op de lokale bodemkwaliteit door het herstellen van het organische stofgehalte.

Deze methode is dus een begin om de potentiële positieve rol te erkennen en te beoordelen die mensen in hun natuurlijke omgeving kunnen spelen.

In Hoofdstuk 5 wordt de volgende onderzoeksvraag beantwoord: *Hoe kunnen de conventionele gecentraliseerde en de gedecentraliseerde benaderingen voor afvalwaterscheidingi worden vergeleken vanuit een water-voedsel-energie perspectief op een geïntegreerde en holistische manier?*

Een nieuw water-voedsel-energie (WFE) nexus-kader wordt gepresenteerd om een conventionele gecentraliseerde RWZI en een gedecentraliseerd gescheiden afvalwater systeem (DSS) te vergelijken. Het bevat indicatoren die de voordelen vastleggen voor de waterzuivering en de voedselproductie sectoren die voortvloeien uit oplossingen voor resource recovery.

Het kader wordt toegepast op twee casestudies die 12.000 i.e. bedie- nen: een conventionele gecentraliseerde RWZI in Corleone, Italië, en een DSS in Helsingborg, Zweden. De Corleone RWZI gebruikt een gecentra- liseerde actiefslib (AS) behandeling met intermitterende beluchting, een oxisch-sedimentatie anaerobe tank, en een ultrafiltratie-eenheid om irrigatie- water te produceren voor boerderijen gelegen op 2 km afstand van de RWZI. Het behandelde afvalwater zal gedeeltelijk worden geloosd in een stroom en gedeeltelijk worden gebruikt voor landbouwirrigatie. Helsingborg scheidt zwart en grijs water met behulp van vacuümtoiletten. Het grijze water wordt getransporteerd met een lagedruk riool en behandeld in een AS-reactor. Het behandelde grijze water wordt geloosd in de oceaan, maar er worden plannen besproken om het te hergebruiken voor de irrigatie van boerderijen. Hier wordt uitgegaan van een gemiddelde afstand van 0,1 km tussen de RWZI en de boerderijen. Het zwarte water wordt anaeroob vergist met het overtollige slib van de AS-eenheid. Het anaeroob digestaat wordt gebruikt voor de productie van bodemverbeteraars en compost voor de landbouw.

Bij vergelijking van de waterzuiveringsprestaties is Corleone beter dan Helsingborg op de meeste efficiëntie- en circulariteitsindicatoren, behalve voor de circulariteit van voedingsstoffen. Dit komt voornamelijk doordat de terugwinningsprocessen in Helsingborg intensiever in hulpbronnen zijn. Helsingborg presteert beter dan Corleone op de meeste efficiëntie- en circulariteitsindicatoren met betrekking tot voedselproductie. Dit is vooral te wijten aan het droge klimaat en de grotere afstand tussen de boerderijen en de RWZI in Corleone. Helsingborg presteert beter dan Corleone op alle natuurreciprociteit indicatoren. Corleone heeft een negatieve zoetwaterherstelwaarde van $-8{,}1\times10^7$ m^3/j, wat betekent dat de verontreinigingen in het effluent, voor hun verdunning, een hogere waterdebiet vereisen dan die van de stroom. Verder behaalt Helsingborg een score van 212% voor energie zelfvoorziening, terwijl de energie zelfvoorziening van Corleone 0% was.

Dit nieuwe kader helpt dus om een multidimensionale vergelijking te maken tussen de twee benaderingen van waterzuivering, rekening houdend met de relevante klimaat- en landbouwomstandigheden. Het gebruik van indicatoren die relevant zijn voor de waterzuivering en de voedselproductiesectoren zal zorgen voor gemakkelijkere communicatie en zou kunnen bijdragen aan een betere coördinatie in de toekomst.

Dit werk levert verschillende wetenschappelijke bijdragen. Ten eerste is de material circularity indicator methode in dit proefschrift uitgebreid om deze geschikt te maken voor zowel de technische als de biogeochemische kringloop. Bovendien biedt de natuurreciprociteit methode een nieuwe visie voor duurzaamheid, kan leiden tot betere beslissingsuitkomsten, en nieuwe paden voor resource recovery onthullen. Ten slotte helpt het nieuwe WFE raamwerk bij het vergelijken van gecentraliseerde en ge-decentraliseerde afvalwaterscheidingsbenaderingen voor en resource recovery op een geïntegreerde manier. De casestudies in dit proefschrift leveren bovendien nuttige gegevens voor toekomstige studies over TW-hergebruik, nieuwe zuiveringsmethoden, en DSS-RWZI's.

Dit onderzoek biedt maatschappelijke bijdragen door implicaties voor besluitvormers te presenteren. De ontwikkelde methoden helpen hen bij het beoordelen van opties voor resource recovery en toepassing, vooral tijdens de planningsfase, en dienen als hulpmiddelen in stakeholder discussies. De circulariteitsmethode stimuleert nadenken over de beste manier om een hulpbron terug te geven aan het milieu, terwijl de natuurreciprociteit methode helpt bij het evalueren van de positieve milieu-impact van teruggewonnen hulpbronnen, wat kan leiden tot andere besluitvorming. Het WFE-raamwerk biedt een praktische manier om conventionele en gedecentraliseerde systemen te vergelijken, rekening houdend met lokale omstandigheden. De toepassing van dit raamwerk zal naar verwachting intersectorale communicatie en coördinatie verbeteren.

Voor toekomstig onderzoek worden de volgende richtingen voorgesteld: het opnemen van actuele emissie- en hulpbronnenwaarden met milieudrempels in LCSA-onderzoek, overstappen op niet-compensatoire aggregatiemethoden voor milieu- en economische criteria, en meer toegankelijke, sectorspecifieke resource flow modellen ontwikkelen. Daarnaast worden potproeven aanbevolen om een database te creëren met diverse landbouwomstandigheden en NUE-waarden voor verschillende nutriëntenproducten. Voor een nauwkeurigere economische efficiëntiebeoordeling is meer data nodig over de kwaliteit en marktprijzen van teruggewonnen producten.

Tot slot kunnen de hier ontwikkelde methoden de planning van resource recovery in de waterzuiveringssector aanzienlijk verbeteren, vooral wanneer ze worden gecombineerd met LCSA-methoden, en houden ze rekening met lokaal relevante factoren.

Preface

It was about 5 years ago that I discovered my love for research and specifically for the field of water management/environmental engineering. The global degradation of water bodies, and the natural environment in general, is a topic that affects me deeply. We have been taking nature for granted and exploiting its resources. I look upon environmental technologies as an intermediary between human society and the natural environment to repair this lopsided relationship. The intermediary can, initially, transform this interaction to be substantially less damaging to nature. But eventually it can also help us develop a reciprocal relationship with nature. Simultaneously, we need some way to assess the performance of these technologies. With this work, I provide methods and metrics to evaluate the environmental technologies in achieving their goal. I hope, with this work, I can make the lives of decision-makers a little easier.

I find it crucial, though, not to over-emphasize the role that environmental technologies have to play in protecting the natural environment. Pollution and resource exploitation are not issues that can be solved using technical solutions alone. As J.B Wiesner and H.F York spoke on the proliferation of nuclear weapons, some issues demand a change in human values and behaviour much more than new technologies. The need to change our values and actions, the kind of change, and the urgency of it are not topics that need to be preached any further. If there ever was a time for action, it is now.

Anurag BHAMBHANI
Delft, August 2024

1

Introduction

1.1. The motivation behind resource recovery

This thesis centers on the water treatment sector's shift towards adopting the circular economy paradigm. This sector can be broadly divided into wastewater treatment plants (WWTPs) and drinking water production plants. The sector meets critical human needs by providing potable water, treating domestic wastewater, and fecal sludge to minimize their contact with humans and the natural environment (Rietveld et al., 2016). However, this is accomplished by using large quantities of chemicals, minerals, and energy, leading to substantial emissions (Morley et al., 2016). To illustrate this point, wastewater treatment is responsible for nearly 3% of the US national electricity consumption (Hao et al., 2019). Additionally, the wastewater conveyance and treatment processes are responsible for 3-7% of the global N_2O emissions (Song et al., 2024), 5-8% of the global methane emissions (Song et al., 2023). Furthermore, the resource consumption and emissions are expected to rise mainly due to a growing demand for drinking water, higher pollutant loads in the wastewater, and more stringent effluent standards (Cardoso et al., 2021). Therefore, the sector has to contend with the pressing need to reduce their resource use and emissions. Consequently, the sector is shifting its focus from standard treatment to prioritizing resource recovery (Van Der Hoek et al., 2016).

Throughout this thesis, 'resource recovery solutions' refers to any process that reclaims, recycles, or reuses the material, or energy that would otherwise be discarded. Recovering resources can reduce the negative environmental effects of treatment plants. For example, resource recovery solutions can decrease the energy footprints, reduce the emissions of greenhouse gases, and lower the eutrophication of a WWTP (Cornejo et al., 2016). The reduced environmental damage may be a result of a lower energy consumption within the WWTPs or due to the avoided burden of extracting virgin resources replaced by the recovered ones. For example, recovery of cellulose in the primary treatment step of a WWTP can reduce the net energy demand of an activated sludge (AS) WWTP by 40% (Ruiken et al., 2013), and nitrogen recovery from domestic wastewater can replace the virgin nitrogen obtained using the energy-intensive Haber Bosch process (van der Hoek et al., 2018).

Resource recovery solutions can also improve the economic efficiency of treatment plants either by reducing costs or generating extra revenue. For example, phosphorus recovery from wastewater as magnesium-ammonium phosphate can improve the dewaterability of digested sludge and thus reduce the sludge disposal costs for a WWTP (Egle et al., 2016).

Another advantage of resource recovery solutions is the benefit that the natural environment can obtain from the recovered resources. The assessment of benefits to the natural environment has not received sufficient attention in literature (Trimmer et al., 2019). Here, this aspect is referred by the term 'nature reciprocity'.

However, none of the benefits mentioned above can be taken for granted since resource recovery does not always lead to a better environmental performance. In fact, nutrient recovery technologies usually lead to a higher global warming impact of WWTPs (Mayer et al., 2021; Pausta et al., 2024; Pradel and Aissani, 2019). This makes it crucial to carefully weigh the cost and benefits of resource recovery solutions.

It is therefore necessary to consider several factors when planning for resource recovery solutions, such as energy, material, and economic efficiencies, circularity, and nature reciprocity. Several resource recovery techniques and the various ways to combine them further make the decision-making process complex (Kehrein et al., 2020), calling for assessment methods that can guide the sector's transition towards the circularity paradigm.

The objective of this thesis is to develop assessment methods to aid decision-makers in the planning and designing of resource recovery solutions linked to the water treatment sector. Four research gaps (RGs) are addressed through the research questions (RQs) presented in the following sections.

1.2. Sustainability assessment of resource recovery

Before discussing any assessment method relevant to the resource recovery solutions, the current literature related to sustainability assessment needs to be consulted. This is because circularity, efficiency, and nature reciprocity are all concepts that can be placed under the umbrella concept of sustainability. As a starting point to the development of the new methods, the strengths and weaknesses are highlighted for the most common sustainability assessment methods employed for resource recovery solutions. These methods together constitute what is called the life cycle sustainability assessment (LCSA) framework.

A framework for sustainability assessment must inform decision-makers on its three pillars: the environment, economy, and society. The predominant framework used for the three-pillar sustainability assessment in an integrated and inter-disciplinary manner is LCSA (Gloria et al., 2017). LCSA comprises of three methods: life cycle assessment (LCA) to evaluate the environmental impacts, life cycle costing (LCC) for the economic analysis, and social life cycle assessment (S-LCA) to assess the social impacts. (Guinée et al., 2011).

As already discussed, recovering resources can help transition the treatment plants towards the circularity paradigm and potentially become more sustainable. Three main characteristics of the resource recovery solutions of the sector can be found in literature. Firstly, recovered resources can positively impact ecosystems, thereby fostering a reciprocal flow of benefits between human 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 exclusively focused on damage reduction and mitigation. Rather, they can also actively benefit the natural environment. It remains to be examined if the methods of the LCSA framework can be used to assess the nature benefits of the resource recovery solutions.

Secondly, water treatment and resource recovery solutions rely on nature. Generally speaking, any economic activity depends on the 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. Similarly, reuse of treated wastewater 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), exhibiting the indispensability of natural resources and processes. An analysis of the relationship between economic development and the natural environment is missing. Also, whether the LCSA framework accurately captures this relationship needs to be evaluated.

Thirdly, resource recovery solutions are motivated by the need to manage the resources better and avoid the transgression of environmental thresholds such as the planetary boundaries (Velenturf and Purnell, 2017; Chrispim et al., 2020). Therefore, preventing the depletion of natural resource and the transgression of the environmental thresholds are central goals associated with the resource recovery solutions. We need to analyse if the environmental thresholds are accounted for by the current sustainability assessment methods in the context of the resource recovery solutions.

While the contribution of LCSA is certainly commendable in promoting life cycle thinking and incorporating environmental, economic, and social dimensions, the following research gap persists.

RG1:- It remains to be examined if the methods of the LCSA framework, as commonly used, are suited to assess the sustainability of the resource recovery solutions.

1.3. Circularity assessment of resource recovery

As already mentioned, the linear economy is unsustainable. Circular economy (CE) is the concept of recirculating resources within the economic system to maximise the recovered value (Corona et al., 2019). The goal of resource recovery is to transition the water treatment sector towards higher circularity and thereby promote sustainability. Since different resource recovery strategies can contribute to resource conservation to varying degrees, assessment methods are needed to select the most effective transition strategy (de Oliveira et al., 2021) and to measure the progress (Saidani et al., 2019).

However, circularity assessment of the water treatment sector can get complex because of a mix of technical (e.g., industrial coagulants) and biogeochemical (e.g., nitrogen) cycle resources. Technical cycle resources are generally abiotic, non-renewable, and synthetic and have the potential to remain circulating in the production system (i.e., industrial manufacturing, recovery, and reuse) (Braungart et al., 2007; Mestre and Cooper, 2017) without being land-filled or used for energy generation (Navare et al., 2021). Biogeochemical resources move in a continuous cycle, passing alternatively between non-living forms and as part of living matter (Bertrand et al., 2015). While most methods are designed for the technical cycle resources, not much research on the assessment of biogeochemical resources exists.

The circularity assessment of biogeochemical resources is not straightforward because of three factors. Firstly, these resources (e.g., water) naturally recirculate (e.g., in the hydrological cycle); hence they can become scarce because the resource may be in a condition that is difficult to use (e.g., water vapour) or accumulate in an environmentally pernicious form (Rijsberman, 2006) (e.g., untreated wastewater).

Secondly, as these resources move through biogeochemical cycles, their various forms fulfill distinct environmental functions (Gleeson et al., 2020; Zipper et al., 2020) (e.g., while the water flowing in an over-land stream sustains aquatic ecosystems, evaporating water helps to cool down the environment). Therefore, simply maximizing a particular resource form for human benefits can disrupt critical ecosystem functions.

Finally, the availability of biogeochemical resources can be significantly affected by the environmental loss mechanisms (Vicente-Serrano et al., 2014) (e.g., evaporation loss of the treated wastewater used for irrigation). These losses have to be determined according to the local conditions; otherwise, the circularity assessment remains superficial. More complications arise because some of these losses on smaller spatial and temporal scales may be beneficial on a larger scale (Grafton et al., 2018). Thus, certain biogeochemical flows categorized as losses can also be considered circular because they enable future resource availability.

Given above, what constitutes circular flows is different for biogeochemical resources than for technical ones, and a different approach to assessing circularity is needed. The current methods, designed for the technical resources, may under- or over- estimate the circularity of resource recovery solutions if applied to biogeochemical resources. Guidance is lacking on how to determine if a certain biogeochemical resource flow can be categorized as linear or circular which leads to the second research gap.

> *RG2:- A method needs to be developed to assess the circularity of resource recovery solutions in the water treatment sector accounting for the complex nature of the biogeochemical resources.*

1.4. Nature reciprocity using resource recovery

Resource recovery from wastewater has the potential to actively provide benefits to nature and assessing these benefits is important (Trimmer et al., 2019). First, this will support a cycle of reciprocal benefits between human society and nature (Trimmer et al., 2019). The natural environment provides numerous services to human society that can be conceptualized using the ecosystem services framework (Wallace, 2007). Human society can also provide benefits to the natural environment. An example of this is the indigenous communities enhancing the soil fertility of the Amazon forests by adding charcoal and bones (Comberti et al., 2015). Mutually beneficial relationships (symbiosis) among organisms and between organisms and their natural environments are common. Yet, human society's consideration of their beneficial role in nature has been limited. The research focus has remained on the reduction of negative impacts.

Second, assessing the nature benefits can provide a more holistic view of wastewater treatment and thereby reveal more resources recovery opportunities (Trimmer et al., 2019). To illustrate, adding organic matter can improve soil structure with secondary benefits such as improved water retention and enhanced vegetative growth. In some contexts, these benefits may be more valuable and should not be ignored in favour of a directly recognizable human benefit, such as energy generation (Trimmer et al., 2019).

Third, only damage reduction is insufficient for sustainability (Hauschild, 2015), especially for the emissions that have crossed the sustainable planetary limits. For example, the anthropogenic emissions of reactive N and P have crossed the limits that planet earth can sustain (Sandström et al., 2023; Steffen et al., 2015). In such cases, an active approach towards repairing the nutrient flows is indispensable.

Trimmer et al. (2019) created a conceptual framework defining the potential pathways for WWTPs to renew ecosystems and highlighting this as an important step for advancing the sustainable development goals. Chrispim et al. (2020) developed an assessment framework specifically for the developing countries. They also assessed the positive environmental effects of resource recovery using a qualitative scale. Kehrein et al. (2020) pointed out that resource recovery can lead to negative and positive environment impacts. They suggest assessing the positive environmental impacts in terms of avoided conventional production of natural gas, cellulose, fertilzers, etc. and also in terms of carbon sequestration. While avoided burden is a way to reduce the negative environmental impact of a WWTP, carbon sequestration can be considered a direct nature benefit.

Despite recognizing the potential for nature's reciprocity and the importance of assessing it, methods that evaluate enhanced ecosystems remain rare (Trimmer et al., 2019) and are often qualitative, resulting in the following research gap.

RG3:- A holistic method for the quantitative assessment of natural environmental benefits is needed.

1.5. Decentralized source separation from a water-food-energy nexus perspective

The resources recovered from water treatment plants are often used in other sectors including food and energy production, and manufacturing industries (Zarei, 2020). These sectors have functions and the related sustainability goals which differ from those of the water treatment sector. For example, the main function of the food production sector is the production of food crops and its sustainability goals could be minimizing the use of freshwater and industrial fertilizers.

The water treatment sector is closely linked to food and energy production, engendering the concept of the water-food-energy (WFE) nexus (El-Gafy, 2017; Molajou et al., 2021). Assessing the sustainability performance of the entire nexus is a complex undertaking (Albrecht et al., 2018; Dargin et al., 2019). While significant work has been done to develop WFE frameworks, a lack of specific and reproducible assessment methods has been pointed out by many (Albrecht et al., 2018; Cairns and Krzywoszynska, 2016; Nhamo et al., 2020; Shannak et al., 2018). Furthermore, WFE frameworks lack the consideration of the local geography, climate, and other consequential factors (Shannak et al., 2018). Therefore, a novel framework is needed to assess the sustainability performance of the WFE nexus.

Furthermore, wastewater treatment and resource recovery can be achieved through conventional centralized treatment, decentralized source separation (DSS) or their combination. Conventional centralized treatment enjoys economies of scale and usually has a lower energy and land use (Besson et al., 2021; Firmansyah et al., 2021; Roefs et al., 2017). DSS refers to a combination of decentralization and source separation wherein domestic wastewater is separated into different streams at the source and the treatment, reuse, or disposal of all or some of the streams are achieved very close to the point of generation. DSS can offer advantages from the resource recovery perspective. Due to the lower dilution of organic matter and nutrients, resource recovery may be more efficient from source-separated streams (Pasciucco et al., 2022). The comparison between the conventional centralized and DSS WWTPs has yielded mixed results depending on the indicators used (Cardoso et al., 2021; Firmansyah et al., 2021; McConville et al., 2017). Furthermore, the two approaches are yet to be compared using a WFE framework to the best of the authors' knowledge. The above analysis leads to the following research gap.

RG4:- An integrated and holistic water-food-energy framework is required to compare the decentralized source separation and the conventional centralized approaches to wastewater treatment and resource recovery.

1.6. Research questions and thesis structure

The aim of this thesis is to develop methods to assess the circularity, efficiency, and nature reciprocity of resource recovery solutions in the water treatment sector. According to the above knowledge gaps, the following research questions are formulated.

> *RQ1:- What are the strengths and weaknesses of life cycle sustainability assessment in the context of resource recovery solutions linked to wastewater treatment plants?*

In Chapter 2, the strengths and weaknesses of the LCSA framework are presented. The chapter provides a starting point to develop new methods that overcome the weaknesses of the LCSA framework and suggests future research directions.

> *RQ2:- How can a method be developed to accurately assess the circularity of the biogeochemical resources present in the water treatment plants?*

Chapter 3 presents a novel circularity assessment method developed specifically for the resource recovery solutions in the water treatment sector.

> *RQ3:- What are the potential nature benefits from the application of the recovered resources and how can they be maximized?*

In Chapter 4, three potential nature benefits are expounded and a novel method is presented to assess the benefits to the natural environment from recovered resources.

> *RQ4:- How can the conventional centralized and the decentralized source separation approaches to wastewater treatment and resource recovery be compared from a water-food-energy nexus perspective in an integrated and holistic manner?*

In Chapter 5, the efficiency, circularity, and nature reciprocity assessment methods are applied to compare two approaches to resource recovery: the conventional centralized WWTP and a decentralized source-separated treatment approach wherein grey water (GW) and black water (BW) are collected separately.

Lastly, some conclusions related to the assessment methods, their utility, and future research recommendations are discussed in Chapter 6. Table 1.1 presents the research questions linked to their relevant chapter and publications.

Table 1.1: The research questions with the corresponding chapters and journal papers.

Research question	Chapter	Journal paper
What are the strengths and weaknesses of life cycle sustainability assessment in the context of resource recovery solutions in the water sector?	2	**Bhambhani, A.,** van der Hoek, J. P., and Kapelan, Z. (2022)., Life cycle sustainability assessment framework for water sector resource recovery solutions: Strengths and weaknesses. *Resources, Conservation and Recycling, 180.*, 10.1016/j.resconrec.2021.106151.
How can a method be developed to accurately assess the circularity of the biogeochemical resources present in the water treatment plants?	3	**Bhambhani, A.,** Kapelan, Z., and van der Hoek, J. P. (2023)., A new approach to circularity assessment for a sustainable water sector: Accounting for environmental functional flows and losses. *Science of The Total Environment, Article 166520.*, 10.1016/j.scitotenv.2023.166520.
What are the potential nature benefits from the application of the recovered resources and how can they be maximized?	4	**Bhambhani, A.,** Jovanovic, O., van Nieuwenhuijzen, A., van der Hoek, J. P., and Kapelan, Z. (2024)., Introducing a new method to assess the benefits of resources recovered from wastewater to the natural environment. *Sustainable Production and Consumption, 46:559–570.*, 10.1016/j.spc.2024.03.016.
How can the conventional centralized and the decentralized source separation treatment approaches be compared from a water-food-energy perspective in an integrated and holistic way?	5	**Bhambhani, A.,** Jovanovic, O., Kjerstadius, H., Di Trapani, D., Mannina, G., van der Hoek, J. P., Kapelan, Z. (2025)., A novel water-food-energy framework for a comprehensive assessment of resource recovery from wastewater treatment plants. *Journal of Cleaner Production, 489, Article 144716.*, 10.1016/j.jclepro.2025.144716.

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2

Life cycle sustainability assessment: Strengths and Weaknesses

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2.1. 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 (Wulf et al., 2019; Purvis et al., 2019) there is some agreement on its three pillars, namely the environment, society, and economy (Purvis et al., 2019; Florindo et al., 2020; Godskesen et al., 2018)

Resource recovery and reuse is one way to promote sustainability of the water treatment sector (Wang et al., 2015). Technological solutions for resource recovery exist, but there is a need for planning and designing methods 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 cost (LCC), and social impacts using social-life cycle assessment (S-LCA) (Guinée et al., 2011). In this chapter, the focus will be on the environmental and the 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 exploring these is beyond the scope of this thesis.

The objective of this chapter is to analyze the strengths and weaknesses of the existing LCSA framework, in the case that it is used to assess the sustainability of the water treatment sector and to suggest modifications, if required. The focus of this chapter will be more on explaining the weaknesses and suggesting improvements. LCSA is analysed using the literature on resource recovery solutions and evaluated by applying it on a real-life case-study.

This chapter is organized as follows. It begins with a section introducing resource recovery solutions and LCSA. A brief description of some characteristics of water treatment sector's resource recovery solutions are presented in Section 2.2. The strengths and weaknesses of LCSA are also described here. Section 2.3 starts with a short description of a case-study and introduces LCA, LCSA, and MCDA as methods used in the case-study evaluation. Section 2.4 presents results obtained in the case-study evaluation for each of the three methods. This is followed by a discussion about the characteristics of LCSA and suggested modifications and future research recommendations using the case-study results as the basis in Section 2.5. Finally, conclusions follow in Section 2.6. The Appendix A includes the LCA template report based on ISO 14040. Supplementary material (S2.1-2.13) contains spreadsheets relevant for the LCA and LCC inputs and their results.

2.2. Background

2.2.1. Resource recovery solutions

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

Firstly, the recovered resources can positively impact ecosystems, thereby fostering a reciprocal flow of benefits between society and nature (Trimmer et al., 2019), e.g., 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 focused on damage reduction and mitigation. Rather, they can also serve as contributors towards benefiting nature.

Secondly, resource recovery solutions rely on nature. Generally speaking, any economic activity, including the 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, the reuse of water for aquifer recharge, which has benefits such as land subsidence prevention, groundwater recharge, etc., depends on the natural filtering capacity of soil (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 the 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 the transgression of environmental thresholds are goals associated with resource recovery solutions.

2.2.2. LCSA of water treatment resource recovery solutions

For appraising sustainability, LCSA uses the LCA, LCC, and S-LCA methods (Costa et al., 2019). The contribution of LCSA is commendable in mainstreaming life cycle thinking and broadening the impacts considered to include the 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 characteristics of LCSA. These are presented under three categories: conceptual, ontological, and methodological.

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 its inclusion of entire life cycles, thus preventing burden shifting (Hauschild et al., 2018). But, it is noteworthy that the concept of eco-efficiency is centred on damage reduction (Barbiroli, 2006; Hauschild and Huijbregts, 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 the author's 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 biological symbiosis, 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 the interactions that mutually benefit the organisms and their natural environments.

Ontological characteristics

The LCSA framework is remarkable for broadening the scope of analysis from only environmental to including the social and economic dimensions (Guinée et al., 2011). This 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).

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 the environmental and the economic indicators in a manner that allows unhindered compensations between them. In other words, a low performance on the environmental criterion can be compensated for by a high economic performance.

The most common approach of aggregating results of an economic and an 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 resilience and economic 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, climate change) can be substituted. 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 environmental damage and economic welfare are treated as entirely substitutable, it effectively implies that economic capital is interchangeable 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, 1992; 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, an infrastructure is needed to capture it, and this requires the natural resources. Therefore, economic welfare cannot be traded with natural capital since the former depends upon the latter. In case the 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. The author contends 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 ecosystem 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 fulfills many functions such as production, pollution absorption, etc. Built capital is usually meant to perform a single anthropocentric function. If built capital were to replace natural capital, many crucial non-anthropocentric functions would no longer be fulfilled.

Methodological characteristics

The 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., 2018).

In LCSA, the method almost exclusively used for the environmental assessment is LCA which measure eco-efficiency i.e., how much resource depletion or emission is caused by a unit operation (Hauschild et al., 2018; 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) presented 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) compared four alternative wastewater treatment pathways concerning carbon emission and eutrophication intensity. They discovered lower carbon emission and eutrophication for decentralized and centralized-decentralized hybrid WWTPs compared to centralized ones. Due to lower emissions, they pointed to the decentralized systems as being more sustainable. To quantify the environmental sustainability of resource recovery solutions, Cornejo et al. (2019) suggested 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 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 by Sandberg et al. (2019). Second, LCA results are usually not linked to environmental thresholds or carrying capacities (Bjørn and Hauschild, 2015). Equating a 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 magnitudes of CO_2 emission.

Secondly, for emissions that may cause irreversible environmental damages, a context-less eco-efficiency comparison of alternatives only determines the alternative that can postpone the threshold transgression longer. To illustrate, continuous P discharge to water bodies can potentially lead to irreparable damage to the aquatic ecosystem (Chowdhury 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 analyzed in the context of the carrying capacity of the receiving environmental compartment.

With continuous eco-efficiency improvements, it may be 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. (2018) 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 (Hauschild et al., 2018). LCA targets the 'T' dimension (Hauschild et al., 2018).

Table 2.1 summarizes the strengths and weaknesses of LCSA, and Section 2.3 will demonstrate these on a case-study.

Table 2.1: Strengths and weaknesses of Life Cycle Sustainability Assessment. This table shows the characteristics of LCSA that may be modified to better assess resource recovery solutions.

Classification	**Strengths**	**Weaknesses**
Conceptual	LCSA avoids burden shifting by including a wide variety of impact categories and analysing complete life cycles.	LCSA focuses 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.	The linear aggregation of results from these three dimensions allows complete compensation between environmental and economic capitals.
Methodological	LCSA enables eco-efficiency improvements.	Absolute sustainability cannot be judged solely with eco-efficiency measurements. Including current emissions and environmental thresholds can make the framework more comprehensive.

2.3. Methods

This section introduces the methods used to evaluate the characteristics of LCSA summarized in Table 2.1. The evaluation will be based on a resource recovery case-study belonging to the water treatment sector.

2.3.1. Case-study description

An Amsterdam-based company is developing a bio-composite 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 ($CaCO_3$), also 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 which leads to a supersaturated condition in the water for calcite. Consequently, calcite crystallizes around a seeding material such as sand. It is also possible to reuse the crystallized calcite as seeding material and create a closed-loop recycling system (Schetters et al., 2015). Only some of the calcite produced by the softening reactors needs to be reused as seeding material; the rest is available as residue. Water reeds, growing naturally along the canals and rivers, are cut and collected regularly. They are cut down to 3–6 mm in length. These raw materials have to be glued together for which resins are used. The manufacturer is contemplating between unsaturated polyester resin and bio-based resins. Polyester resin is assumed in this case-study.

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 600×600×10.5 mm. The boards are expected to be used as construction material for river/canal bank protection. Figure 2.1 shows a flowchart describing the various processes of the bio-composite life cycle.

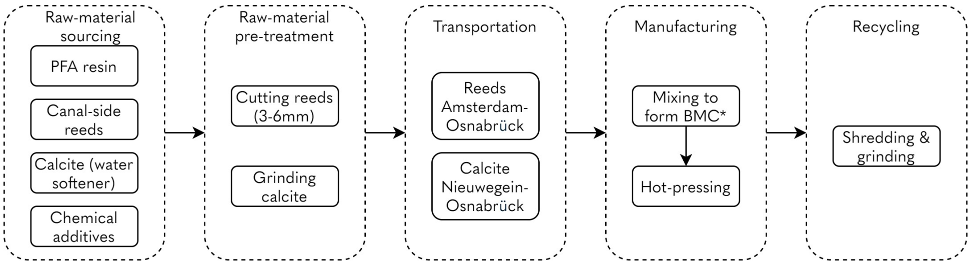

Figure 2.1: Life cycle stages of the bio-composite: raw-material sourcing, raw material pre-treatment, transportation, manufacturing, and recycling. *Bulk moulding compound

2.3.2. Life cycle assessment (LCA)

An LCA is conducted on the material described in Section 2.3.1. The ISO 14040 template is followed, which consists of goal and scope definitions, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation. Here, only those sections are shown that are needed to understand the goal and scope and are directly relevant to the discussion of LCSA.

Goal definition

The LCA will reveal environmental impacts associated with the production, use, and recycling of the bio-composite material and identify environmental hot-spots. The study targets the bio-composite manufacturing company, water treatment facilities, and academics. This LCA is primarily to analyse the use of the LCSA framework in water sector resource recovery solutions and demonstrate its characteristics. Hence, the outcomes of should be seen as indicative and can serve as a basis for a more detailed analysis.

Scope definition

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. Figure 2.2 shows the system flowchart.

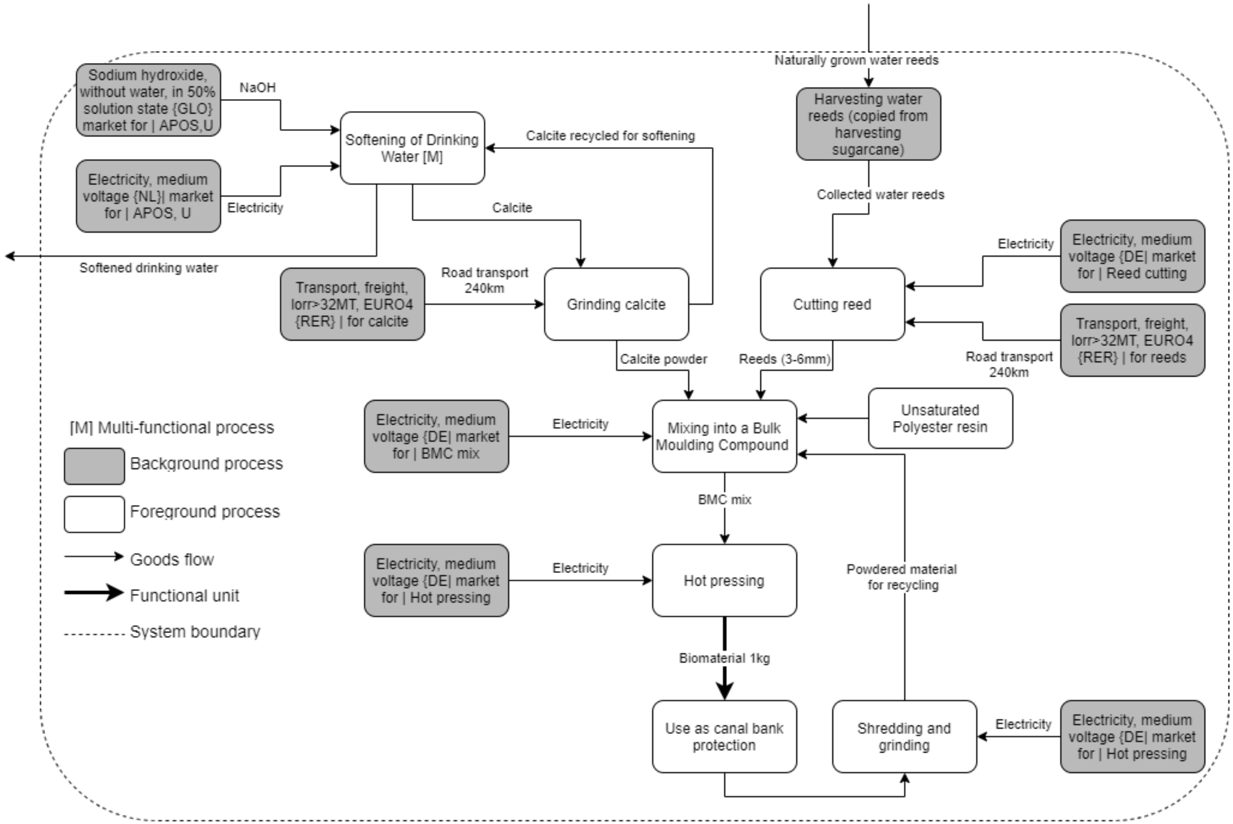

Figure 2.2: A flowchart of the cradle-to-cradle LCA of the bio-composite's application as canal bank protection. The system boundary begins with raw material extraction from the urban water system. The bio-composite is used as canal bank protection. The last process is shredding and grinding the material to use as feedstock for new production.

2

Through an LCA, the 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 100×100×2000 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 600×600×10.5 mm, which is the reference flow.

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. The details of this process are presented 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 1.46 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 ± 10 g C/m^2. For the complete LCI, Supplementary material S2.1 may be consulted.

2.3.3. Life cyle costing (LCC)

Life Cycle Costing (LCC) is a method to estimate the total cost 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 the bio-composite manufacturer. The costs are only estimations, and strong conclusions regarding the actual LCC of the material are not recommended.

2.3.4. Multi-criteria decision analysis (MCDA)

Multi-criteria decision analysis (MCDA) methods combine the results from multiple dimensions of sustainability. 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 bio-composites 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). The 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 S2.10.

A linear additive aggregation shown in Equation 2.1 is used to demonstrate the most common MCDA approach that provides a single composite indicator. 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 sustainability.

$$CI = ED + EC \tag{2.1}$$

2.4. Results

2.4.1. Life cycle impact assessment

Characterization

Life cycle inventory of elementary flows is converted into environmental impacts through characterization factors in life cycle impact assessment. In this section, the characterized environmental impacts are presented. In Table 2.2, only a few selected impact categories are shown that will be used in our discussion on some critical characteristics of LCSA .

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., 2018). For the ReCiPe 2016 method, normalization factors were introduced by the method developers in 2010, to normalize the 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 2.3 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 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 the 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.

Table 2.2: LCIA results for the bio-composite application to canal bank protection. The table shows impact categories, their units, values, the primary responsible substance and activity from the life cycle.

Impact category	Unit	Impact	Major elementary flow contributor (compartment)	Major activity contributor
Global warming	kg CO_2 eq.	8.77×10^{1}	CO_2, fossil (air)	82% Polyester resin production
Freshwater eutrophication	kg P eq.	3.62×10^{-2}	Phosphate (water)	43% Polyester resin production, 36% Shredding and grinding for recycling
Terrestrial ecotoxicity	kg 1,4-DCB	2.22×10^{2}	Copper (air)	86% Polyester resin production
Freshwater ecotoxicity	kg 1,4-DCB	3.70	Copper (water)	78% Polyester resin production
Marine ecotoxicity	kg 1,4-DCB	4.84	Copper (water)	77% Polyester resin production
Human ecotoxicity	kg 1,4-DCB	8.00×10^{1}	Chromium VI/Zinc (water)	68% Polyester resin production 19% Shredding and grinding for recycling

Table 2.3: The seven highest environmental impacts based on normalization to global average 2010 per-capita impacts.

Impact categories	Normalized impact (Person.year (2010 World)
Marine ecotoxicity	4.69
Freshwater ecotoxicity	3.02
Human carcinogenic toxicity	1.04
Human non-carcinogenic toxicity	5.17×10^{-1}
Terrestrial ecotoxicity	2.14×10^{-1}
Freshwater eutrophication	5.58×10^{-2}
Global warming	1.10×10^{-2}

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.

Contribution analysis

The purpose of contribution analysis is to identify the contribution of different activities from the life cycle towards an environmental impact. In Figure 2.3, the contribution analysis results are shown 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 the Supplementary material S2.8.

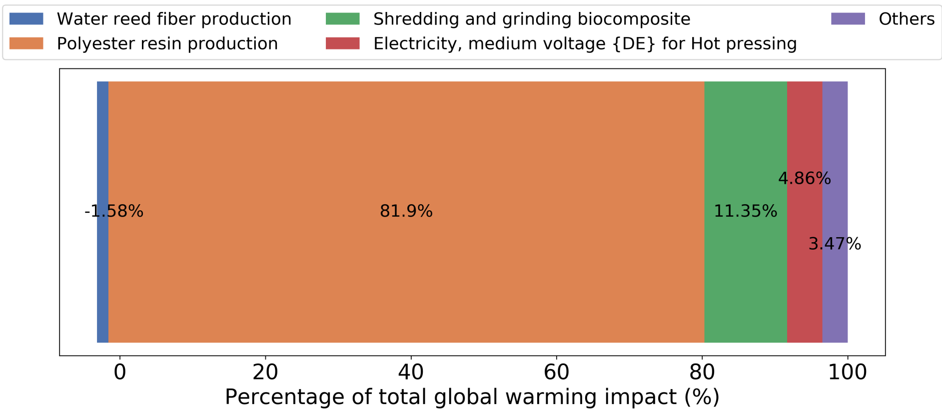

Figure 2.3: Global warming impact contribution from parts of the bio-composite life cycle. All the processes with contributions less than 1% each have been clubbed under 'others' for ease of depiction on the graph.

Comparison with an alternative solution

An assessment of hardwood as an alternative canal bank protection material was conducted to observe how the LCA results can be used when multiple alternatives are compared. LCIA results for the hardwood can be found in Supplementary material S2.11. The 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.

2.4.2. Life cycle costing

The costs associated with 6.96 standard boards are presented in Table 2.4. As shown, the total life cycle cost adds up to €201.0 for 6.96 standard boards. It must be noted that discounting of future costs was not done for simplicity.

Table 2.4: Life cycle cost of 6.96 standard bio-composite boards.

Cost item	**Cost (€)**
Raw material (purchase and pre-treatment	10.5
Transportation of raw material	3.4
Manufacturing (energy and material)	178.2
Installation at canal side	4.4
Recovery and recycling	4.5
Total cost	**201.0**

2.4.3. 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 €21.2. For the complete calculation, S2.12 may be consulted. Environmental damage shadow prices and economic cost obtained earlier are added to express sustainability in a single composite indicator as in Equation 2.2:

$$CI = 201.0 + 21.2 = €222.2 \tag{2.2}$$

A total aggregated score of €222.2 is obtained, combining un-discounted 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? The next section continues the 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.

2.5. Discussion

2.5.1. Methodological characteristics of LCSA

Earlier, a methodological characteristic of LCSA was discussed: equating a context-less eco-efficiency with environmental sustainability. The case-study demonstrates that an 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 the case-study, the ecotoxicities were found to be the most significant environmental impacts. Next, it reveals that polyester resin manufacturing is the most environmentally damaging process or the hot-spot in the life cycle. Information about the most significant impacts and hot-spots can help improve the 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., 2018; Pelletier et al., 2019). Moreover, resource recovery solutions are meant to manage resources better and avoid the transgression of environmental thresholds.

Equating eco-efficiency with sustainability leads to two problems. 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 2.2), current emissions and emission thresholds should be established. To demonstrate one way of doing this, the total GHG emissions was calculated 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 (European Commission, 2017), 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 2.45×10^{-3} t CO_2 eq./capita/year. The current GHG emissions attributed to the Netherlands are 11.4 t CO_2 eq./capita/year (CBS, 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 the remaining budget of GHG emissions is included. There are several ways to calculate the emission budget of a nation. Here, the method described by Romanovskaya and Federici (2019) is used 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 2.45×10^{-3} 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 2.45×10^{-3} 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 the 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 2.4.1, 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 the 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.

2.5.2. Conceptual characteristics 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 2.2, it is clear that the LCSA approach to environmental sustainability is one of damage reduction. For example, through an LCA, an eutrophication damage of 3.62×10^{-2} 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 chapter 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. To explain the nature reciprocity component of this 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). This model will be used 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 the biosphere or the geosphere to the atmosphere (Ajani et al., 2013). A negative emission, crucially, helps to transfer excess CO_2 out of the atmosphere. 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 the impact categories that have crossed or neared environmental thresholds. To differentiate from emission reduction, the sub-concept of 'Reciprocity' is here introduced 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 for two reasons.

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 Figure 2.3.

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 the 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.

Thus, it is not enough to minimize damages caused to nature but it is necessary to actively restore its capacity to support human life. 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.

2.5.3. Ontological characteristics 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 2.5. 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 €73.3. A simple LCA was conducted for hardwood application for canal bank protection, and results were converted into an external cost value of €90.2. A linear additive aggregation provides a composite indicator value of €163.5, a lower and, thus, more sustainable value compared to the bio-composite (€222.2). 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, three arguments for why weak sustainability is unfounded were presented. Firstly, all 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 the growth of raw materials such as water reeds. For instance, the natural cycles of carbon and nutrients like phosphorus and nitrogen are essential 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, it was 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, 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. Goodland and Daly (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.

2.5.4. Future research directions

From the 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, three future research directions are proposed.

Firstly, characterized LCA results, as shown in Table 2.3, 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 treatment sector. The method proposed by Romanovskaya and Federici (2019) was demonstrated 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 treatment sector need defining. This helps link characterized results of LCA to absolute limits and ascertain if a resource recovery solution allows the sector to actually avoid the transgression of environmental thresholds and for how long.

Secondly, 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., may be incorporated into assessment frameworks. The effect of a resource recovery solution on the stocks and flows of these cycles may 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 to provide details such as duration, quantity, and quality (biogenic or fossil) of carbon sequestration, which are critical distinctions (Ajani et al., 2013). This would make it easier to conceptualize proactive solutions for redistributing carbon out of the atmosphere. Research is also needed to develop indicators to quantify improvements that a resource recovery process has on the 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, any assessment framework must avoid 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, 1997), 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.

2.6. Conclusions

The aim of this chapter was to analyse the strengths and especially the weaknesses of the existing LCSA framework, in the case that it is used to assess the sustainability of resource recovery solutions and to suggest necessary modifications. Three characteristics of LCSA were identified that could be modified to serve the 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. First, the strengths of LCSA are breifly summarized as follows:

- Conceptually, LCSA avoids burden shifting by assessing a complete life cycle using a comprehensive selection of environmental impact categories.
- Ontologically, it assesses the environmental, economic, and societal dimensions and can reveal the trade-offs between them.
- Methodologically, it can help to improve the eco-efficiency of a process by revealing environmental hotspots.

The three LCSA characteristics that can be modified and the corresponding research directions are as follows:

- Conceptually, the LCSA framework remains focused on assessing negative impacts of humans on nature. The positive impacts of resource recovery solutions are expressed in terms of negative emission, hiding crucial details about the dynamics of the 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 methods and indicators to assess reciprocal benefit from humans to nature in detail need to be developed.
- A methodological type limitation of the existing LCSA framework is the use of LCA without using thresholds and current emissions/resource stocks. Expressing sustainability in terms of emissions or resource consumption alone can potentially underestimate environmental issues and possibly neglect the environmental threshold transgressions.
- Eco-efficiency assessment should be combined with assigning a 'safe operating space' to resource recovery solutions by scaling down the remaining emission/resource budgets.
- From an ontological point of view, LCSA 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.

- Methods to limit the extent of this substitution 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.

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3

Circularity assessment: Environmental losses and functional flows

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3.1. Introduction

Growing resource use intensity and waste production are causing scarcity of resources and environmental pollution. Therefore, reducing the reliance on virgin resources and avoiding their dissipation are fundamental for sustainability, and this is well understood in the water sector. Consequently, concepts such as carbon neutrality (Mo and Zhang, 2012), nutrient recovery (Mo and Zhang, 2013), and wastewater reuse (Lyu et al., 2016) have been explored in recent years. The strategies to recover resources from the urban water cycle can be broadly classified as resource recovery solutions. As a part of the circular economy, they are meant to decouple economic development from resource extraction by recirculating resources (Corona et al., 2019) and thereby contribute towards a sustainability transition.

Different resource recovery strategies can contribute to resource conservation to varying degrees. Therefore, assessment methods are needed to select the most effective circularity transition strategy (de Oliveira et al., 2021) and to measure the progress towards the circular economy (Saidani et al., 2019). Since the water treatment sector deals with a mix of technical and biogeochemical resources, the circularity assessment method must be sophisticated enough to capture the complexities of these resource flows. While most assessment methods are designed for the technical cycle resources, not much research has gone into the circularity assessment of biogeochemical resources. Current methods designed for the technical cycle resources may under- or over- estimate the circularity of resource recovery solutions if applied to biogeochemical resources. For the circular economy to support sustainability, the definition of the circularity of all resources should be aligned with sustainability.

This study aims to develop a new and improved approach to the circularity assessment of biogeochemical resources commonly found in the water sector in two steps. Firstly, a method is introduced for segregating biogeochemical flows as linear or circular that ensures that the resource flows serving environmental functions are counted towards improved circularity. Also, very few studies base their circularity assessment on resource flow models accounting for the local conditions. Here, a more realistic circularity assessment of biogeochemical resources is achieved by basing the assessment on a resource flow model that considers the local climate and the resource application schedule.

Section 3.2 starts with the definitions of circularity and sustainability, and highlights that circularity should be assessed in a way that can support sustainability. Next, some circularity assessment methods are discussed in Section 3.2 to indicate that they are mainly suited for the technical cycle resources. Thereafter, three factors related to the biogeochemical resources are presented that make defining and assessing their circularity more complicated than doing so for the technical cycle resources. Also, in Section 3.2, the existing material circularity indicator (MCI) method is described, the restorative, regenerative, and linear flows, as originally introduced by the Ellen MacArthur Foundation (2019) as part of their MCI method, are presented, and also the lack of consensus on how to define these terms is discussed. Next, new definitions for the three terms (linear, restorative, and regenerative flows) and the new assessment approach of the MCI method are presented in Section 3.3. Next, the new circularity assessment approach is demonstrated on a real-life case-study involving treated wastewater fertigation. Section 3.4 presents the circularity assessment results. This is followed by a discussion of the case-study results in Section 3.5, where factors that improve water and nitrogen circularity in fertigation are analysed. Subsequently, the differences between the new assessment approach and the original MCI are presented. Finally, the conclusions about the new circularity assessment method and the fertigation case-study follow in Section 3.6. Supplementary material (S3.1-3.9) show the background calculations.

3.2. Background

3.2.1. Circularity and sustainability

The concepts of sustainability and circularity are introduced here not to discuss the details of their various definitions but, to support the opinion of the authors that circularity should be defined and assessed in a way that supports sustainability. This is to avoid ignoring the wider environmental implications of the resource recovery solutions and thereby propagating circularity for circularity's sake (Harris et al., 2021).

Several definitions of sustainability exist, but the most popular one is based on Brundtland (1987): economic development that meets current needs without compromising the needs of future generations. Conserving natural resources for future generations is thus essential for sustainability, and this is where the circular economy fits in.

The circular economy is a concept of recirculating resources within the economic system to maximize the value recovered from them. This concept has been developed as an alternative to the linear economy, where resources are extracted, used, and discarded as waste (Corona et al., 2019). The goal of resource recovery solutions is to decouple economic development from virgin resource extraction and thereby promote sustainability (Bhambhani et al., 2022). Although circularity is meant to promote sustainability (Corona et al., 2019; Terra dos Santos et al., 2022), it may not always do so (Mancini and Raggi, 2021). Thus, it is crucial to assess circularity in a way that supports sustainability (Harris et al., 2021).

3.2.2. Circularity assessment

Since a method is needed to assess the water sector's circularity transition, some of the current methods were analyzed. Since most methods are focused on the technical cycle (Navare et al., 2021; Rocchi et al., 2021), these are discussed first. Collection rates (Haupt and Hellweg, 2019), the percentage of a resource collected after use for recycling, may be used to measure circularity. The recycling rate (Haupt and Hellweg, 2019) is another similar indicator representing the fraction of resources that becomes part of a secondary product. Thus, higher collection and recycling rates may imply higher circularity. In the case of using treated wastewater for irrigation, even though avoiding freshwater will lead to improved circularity, only a tiny percentage of the irrigation water becomes part of the crop (secondary product). A large portion is evaporated, transpired, or seeps underground, leading to a low recycling rate yet contributing to groundwater recharge and thus water sustainability (Kazem Attar et al., 2020).

The circular economy index method (Di Maio and Rem, 2015) assesses circularity as the ratio between the market economic value produced by a recycler to the material economic value entering a recycling facility. This is fine from an economic point of view, but a method solely based on economic value maximization might lead to the biogeochemical resources being diverted towards activities that generate the highest economic returns even at the cost of maintaining environmental functions (e.g., improving irrigation efficiency at the cost of groundwater recharge). The circularity indicator developed by Franklin-Johnson et al. (2016) measures the time duration of resource use, focussing on ‘materials moving perpetually within industrial systems’ (Franklin-Johnson et al., 2016). This indicator maximizes resource access for human functions which can lead to undesirable consequences. This is demonstrated by the fact that maximizing treated wastewater reuse at the cost of reduced discharge into a stream is known to cause a reduction of stream flow quantity and degradation of the stream water quality (Wolfand et al., 2022).

Next, the methods directly relevant to the water sector are discussed. While several water balance studies exist, including Kenway et al. (2011), Currie et al. (2017), and Venkatesh et al. (2017), circularity assessment of water and the resources present in water has received very little attention (Arora et al., 2022; Renfrew et al., 2022). Preisner et al. (2022) compiled a set of indicators for the circularity assessment of the water sector. They proposed a method using the average of the recovery rates of nutrients, and organic matter, the reuse rate of treated wastewater, and the energy sufficiency of a WWTP. This indicator is relatively simple to calculate and can help summarize the WWTP performance in recovering important resources. But, the application scope of this indicator is limited to a WWTP and does not include the resource application process (e.g., irrigation). Kakwani and Kalbar (2022) have developed the water circularity indicator (WCI) based on the MCI method developed by the Ellen MacArthur Foundation (2019). A city-wide urban water circularity framework has been developed by Arora et al. (2022). Both approaches help to assess the urban water systems but exclude hydrological flows such as evaporation, transpiration, runoff, and infiltration losses. Nearly 70% of the total water used by humans is for agricultural irrigation (Cassardo and Jones, 2011), and flows such as evapotranspiration, runoff, and infiltration constitute a large part of the agricultural water flows (Kazem Attar et al., 2020). Therefore, a discussion needs to be started about modelling these environmental losses and assessing their effect on circularity.

Based on the discussion above, three observations about the current assessment methods are presented. Firstly, with these methods, one assumes that retaining resources for human use alone (through high collection rates, high recycling rates, and lengthening the use duration) constitutes circularity, i.e., resource availability for human functions is maximized while every other flow is considered as a 'waste'. This may be appropriate for the technical cycle resources for which defining 'waste' flows such as landfills is straightforward. However, for the biogeochemical resources, 'waste' flows may also be beneficial as long as they contribute towards the sustainability of a resource. For example, irrigation water leakage can contribute to a large portion of groundwater recharge in some regions (Bouimouass et al., 2020; Qi et al., 2023).

Secondly, using the existing methods that maximize the human functions of resources can backfire and lead to resource scarcity. To illustrate, maximizing agricultural water efficiency by reducing groundwater infiltration is known to contribute to groundwater scarcity and reduce environmental flows necessary for sustaining aquatic ecosystems (Batchelor et al., 2014; Simons et al., 2020).

Thirdly, when using the existing assessment methods, one cannot account for the environmental losses of biogeochemical resources, a factor that substantially affects circularity. For example, nitrogen recovered from wastewater can be applied to agricultural soil as a recycled fertilizer. However, the recycled nitrogen that leaches into groundwater cannot be considered circular because this will contaminate the groundwater. And the amount of nitrogen leaching strongly depends on factors such as climate, precipitation (Jabloun et al., 2015), and application rate (Bowles et al., 2018; Shepherd, 1996). Therefore, modeling these factors is crucial for an accurate assessment and the the discussed methods do not take into account such complexities of assessing the circularity of biogeochemical resources.

3.2.3. Biogeochemical and technical resources

Resources can be categorized as technical and biological (Moreno et al., 2016). The technical resources are generally abiotic, non-renewable, and synthetic and have the potential to remain circulating in the technical cycle (i.e., industrial manufacturing, recovery, and reuse) (Braungart et al., 2007; Ellen MacArthur Foundation, 2019; Mestre and Cooper, 2017), without being disposed in landfills or used as fuel for energy generation (Navare et al., 2021). For example, metals need to be reused as many times as possible to avoid their disposal in nature (Velenturf et al., 2019). In contrast, biological resources can safely cycle between the technical cycle and the natural environment (Braungart et al., 2007; Moreno et al., 2016).

'Biogeochemical' is a better term to describe those resources that can cycle between the technosphere and the natural environment since these resources need not always be of biological origin. Biogeochemical resources are those resources that move in a continuous cycle, passing alternatively between a non-living form and as part of living organisms (Bertrand et al., 2015). This study is concerned with the short–term biogeochemical cycles with a time scale of a few years, such as the short-term carbon or nitrogen cycle. For the same reason, phosphorus is not included because it is mostly mined from non-renewable phosphate rocks (Liu et al., 2023; Scholz et al., 2013). The method developed here can nonetheless also be applied to non-renewable resources such as phosphorus.

There are three main differences between the two resource categories that are relevant to circularity. Firstly, differentiating between the linear and circular flows for biogeochemical resources is more complex. Landfilling or energy recovery are linear flows for technical resources because they render these resources permanently unavailable for future use. In contrast, biogeochemical resources naturally flow in short-term cycles (i.e., several years), making it more complicated to differentiate between their linear and circular flows. For example, releasing inert N_2 into the atmosphere can be considered a waste flow (van der Hoek et al., 2018), but simultaneously, closing the nitrogen cycle (by releasing N_2 back into the atmosphere) is considered a pathway for indirect reuse of nitrogen (Spiller et al., 2022) and a way to remove excess reactive nitrogen from the environment (Galloway et al., 2021). So, are dinitrogen emissions from the WWTP linear or circular flows?

Secondly, unlike technical cycle resources, biogeochemical resources serve both human and environmental functions. For instance, treated wastewater (TW) discharge is beneficial for sustaining stream flows and the ecosystems linked to it (Rice and Westerho, 2017) (environmental function), but the TW may be diverted for irrigation (human function), leading to a stream flow reduction and potential ecosystem losses (Rice and Westerho, 2017). So does complete water circularity mean maximizing water use for human benefits at the cost of the ecosystems?

Thirdly, environmental loss mechanisms can be a significant cause of biogeochemical resource scarcity, despite recycling. For example, recovered nitrogen used in agriculture may be dissipated in reactive forms causing substantial economic and environmental damage (Müller and Clough, 2014). Similarly, rising evaporative water loss can be a cause of water scarcity (Vicente-Serrano et al., 2014). Such environmental losses depend on the local climate, soil characteristics, rainfall, etc. Furthermore, some losses may even contribute to circularity. For instance, groundwater infiltration is considered to be irrigation loss at the farm scale, but this 'loss' may simultaneously recharge the groundwater storage (Grafton et al., 2018; Kazem Attar et al., 2020). So, should infiltration loss of water be considered a linear flow even though it may prevent future water scarcity? To assess the circularity of biogeochemical resources in a way that supports sustainability, the complexities of the biogeochemical cycles have to be considered. Figure 3.1 shows a typical technical resource cycle and the nitrogen cycle as an example of a biogeochemical resource to illustrate how much more complicated the flows of the latter can be.

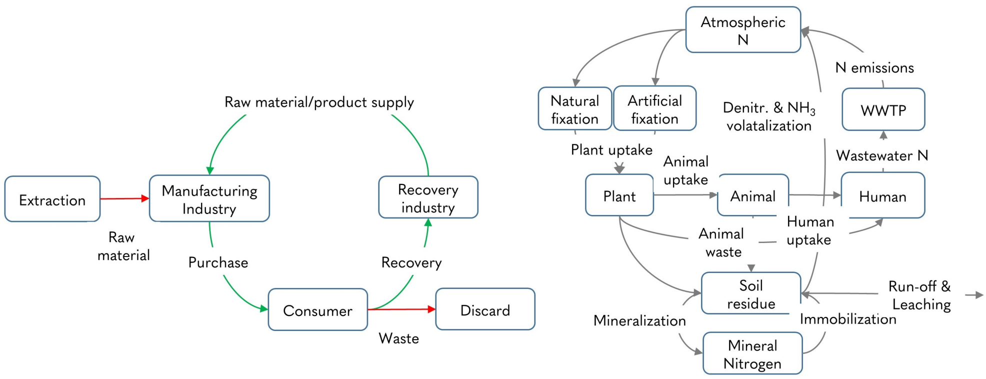

Figure 3.1: (a) A typical technical resource cycle for which it is relatively simple to differentiate between circular (green) and linear flows (red); (b) The nitrogen cycle makes it complex to segregate circular and linear nitrogen flows.

3.2.4. The material circularity indicator

Out of the circularity assessment methods discussed in Section 3.2, the MCI is the most promising for the water sector because of two reasons. Firstly, the MCI covers the input as well as the output circularity of a process. Therefore, one can assess the percentage of the resource feedstock as well as products that can be considered circular. Secondly, the MCI relies on readily-available mass or volume data. Thus, as also considered by Kakwani and Kalbar (2022), MCI provides a good starting point for developing a novel circularity assessment method for the water sector. Still, it is only a starting point as the MCI method too has limitations to be addressed.

The MCI method is based on differentiating restorative and regenerative flows from linear flows. Restorative flows are defined as those that are reused/recycled, and linear flows are the ones that originate from virgin sources, ending up in landfills or energy recovery processes (Ellen MacArthur Foundation, 2019). Regeneration refers to the returning of biotic resources to the natural environment such that the resources remain biologically accessible and the production capacity of the natural source is maintained (Ellen MacArthur Foundation, 2019). However, these definitions do not help distinguish between linear and restorative/regenerative biogeochemical flows. For example, recovered nitrogen applied to the soil may leach into groundwater which is a linear flow even though it is not landfilled nor used for energy recovery. Thus, direct application of the MCI method to the water sector resource recovery solutions is problematic because this restricted way of defining linear and restorative/regenerative flows does not apply to most biogeochemical resources.

Original MCI method

Below, the first two steps of the MCI method are shown to calculate the virgin resource input and the unrecovered waste of a process, and then these steps are used as a framework for calculating biogeochemical resource circularity.

1. Calculate virgin resource input as follows:

$$V = M(1 - F_{\mathrm{R}} - F_{\mathrm{U}} - F_{\mathrm{S}}) \tag{3.1}$$

   where V is the virgin resource input, M is the total resource input, F_R is the feedstock fraction derived from recycled sources, F_U is the feedstock fraction derived from reused sources, and F_S is the fraction of biological resources obtained from sustained production.

2. Calculate the un-recovered waste output as follows:

$$W = M(1 - C_{\mathrm{R}} - C_{\mathrm{U}} - C_{\mathrm{C}} - C_{\mathrm{E}}) \tag{3.2}$$

   where M is the total resource input, C_R is the fraction of the resource flowing into a recycling process, C_U is the fraction of the resource flowing into component reuse processes after the use phase, C_C is the fraction composted, and C_E is the fraction originating from sustained biological production and used for energy recovery.

As can be seen from Equations 3.1 and 3.2, the MCI requires the estimation of linear flows by subtracting the regenerative/restorative flows from the total resource throughput. Resources that originate from reuse/ recycle sources or from sustained biological production are considered restorative/regenerative. On the other hand, all resource outputs that go into a reuse/recycle process, are composted, or are used for energy recovery are considered restorative/regenerative. These definitions of regenerative/restorative flows work well for technical resources but not for biogeochemical flows. For example, WWTPs denitrify nitrogen oxides to emit nitrogen gas (N_2) and nitrous oxide (N_2O) (in case of incomplete nitrification or denitrification). Should inert N_2 emissions be considered linear only because they are not reused/recycled? Clear guidance is lacking that would help to classify biogeochemical flows as regenerative or linear.

3.3. Methods

3.3.1. Redefining restorative, regenerative, and linear flows

Morseletto (2020) has pointed to the lack of a clear definition for the terms 'restoration' and 'regeneration' in the circular economy literature and has defined the term 'restoration' as a return to a previous or original state. Morseletto (2020) also proposes a definition for 'regeneration' as aiding the self-renewal capacity of natural systems against overexploitation by humans. But, the authors also suggest omitting the concept of 'regeneration' as a central circular economy principle due to a lack of robust guidance on how to apply this concept to resource recovery solutions. Since returning a resource to a previous state for human use and releasing the resource into nature are two very different kinds of flows, the term 'regeneration' is retained here as a circular economy principle. The terms restorative, regenerative, and linear flows are here redefined by the authors:

Restorative flow is a flow that recovers a resource for direct human use (e.g., recovery of the struvite fertilizer out of wastewater through precipitation).

Regenerative flow is a flow that returns a resource to the state in which it was originally appropriated from nature for human use. This is to promote the self-renewal and ecosystem-sustaining capacity biogeochemical cycles in response to overexploitation (e.g., releasing reactive nitrogen as N_2 into the atmosphere to close the nitrogen cycle).

Linear flow is a flow that is obtained from virgin sources and/or discarded in a form different from how the resource was originally obtained for human use (e.g., returning water obtained from a river as water vapour to the atmosphere).

These definitions can now be used to differentiate between restorative, regenerative, and linear biogeochemical flows. For example, nitrogen (N_2) is converted into biologically active forms through the Haber Bosch process (Razon, 2018), and if this nitrogen is returned to the atmosphere as N_2, then the return flow can be considered regenerative. On the contrary, water obtained from a freshwater source in liquid form, used for irrigation and returned to nature as water vapour is a linear flow. While it is true, that any water body exposed to the atmosphere will have some evaporation but this is usually a natural process and of a much smaller magnitude compared to the evaporation from an irrigated field. A concern may be raised about the high energy use of obtaining reactive nitrogen from the atmosphere. Energy use is an inevitable factor to be considered for the sustainability of a process. However, the concept of energy use should not be mixed up with circularity. Often high circularity comes at the cost of high energy (Campbell-Johnston et al., 2019; Gregson et al., 2015). Even certain nitrogen recovery technologies, such as air stripping, can have an energy consumption in the same order as required for fixing atmospheric nitrogen using the Haber Bosch process and converting back to N_2 using nitrification-denitrification (van der Hoek et al., 2018).

New material circularity assessment approach
The circularities of the biogeochemical and technical resources are assessed using the following equations:

1. Calculate the virgin inputs as follows:

$$V = M(1 - RSIF) \tag{3.3}$$

   where V is the virgin resource input, M is the total resource input, and RSIF is the restorative input fraction comprised of the input resource that originates from the same or another use process.

2. Calculate the virgin inputs as follows:

$$W = M(1 - RSOF - RGOF) \tag{3.4}$$

   where W is the total unrecoverable waste, M is the total resource input, RSOF is the restorative output fraction defined as accumulated resource fraction (below saturation) or modified to a previous state for human functions, and RGOF is the regenerative output fraction defined as the resource fraction modified to the original state in which these were obtained from nature.

3. Calculate the virgin inputs as follows:

$$LFI = \frac{V + M}{2M} \tag{3.5}$$

   where LFI is the linear flow indicator, V is the virgin resource input (Equation 3.3), W is the unrecoverable waste output (Equation 3.4), and M is the total resource input.

4. Calculate the virgin inputs as follows:

$$MCI = (1 - LFI) \times 100 \tag{3.6}$$

   where MCI (%) is the material circularity indicator of a resource, and LFI is the linear flow indicator of the resource (Equation 3.5).

In the original MCI method, a utility factor (F(X)) is used to penalize for potentially lower product durability resulting from recycled ingredients. However, the industrial analogy of product durability does not apply to the water sector (Kakwani and Kalbar, 2022), and hence, the utility factor is excluded from the MCI calculation. In the next section, the new circularity assessment method is demonstrated in a case-study. The new MCI values are generated and compared to the values of the original MCI method.

3.3.2. Case-study

Description

A block scheme of the Corleone case-study in Italy is shown in Figure 3.2. An activated sludge WWTP treats 3700 m^3/d of domestic wastewater (Mannina et al., 2022). A portion of the treated wastewater (TW) will pass through an ultrafiltration unit which is added to the existing WWTP to produce irrigation water for agriculture (Mannina et al., 2022). The system boundary starts at the point from where the irrigation water is sourced for irrigation and ends at the agricultural land. For assessing a resource recovery solution, the WWTP should also be included within the system boundary. But, in this demonstration, the choice of a narrower system boundary is inspired by two reasons. Firstly, the difference in the assessment results between the original MCI and the modified approach will show up in the irrigation process because of environmental losses such as groundwater infiltration. Such losses do not form any significant part of the flows through the WWTP, and thus, the difference in the assessment results will not be substantial. Secondly, since the focus is on the demonstration of the new method, including only the irrigation process allows for simplicity without losing any generalizability.

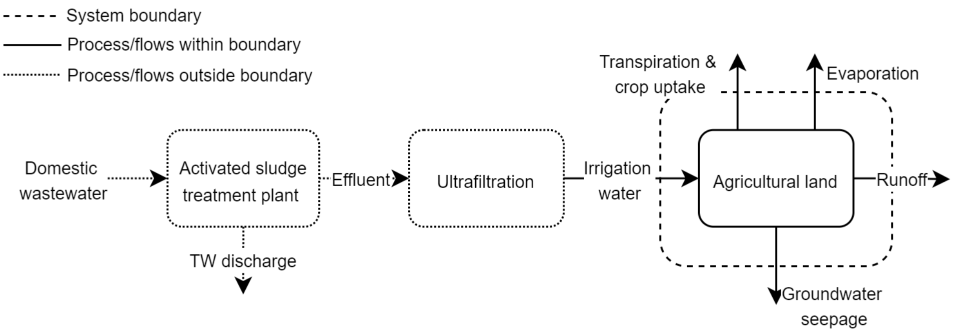

Figure 3.2: Block scheme for reuse of treated wastewater (TW) for fertigation in Corleone, Italy. The circularity for the irrigation process is assessed.

Water and nitrogen circularity assessment

The circularity is assessed over the crop growing period within a year. First, the circularity of irrigating every three days using freshwater (FW_3) is compared to using treated wastewater (TW_3). Next, the circularities of fertigation every three (TW_3) and ten days (TW_10) are compared since the schedule is an important factor affecting the water and nutrient balances in agriculture (Mermoud et al., 2005). The recommended irrigation schedule for the tomato crop is highly location dependent and can range from every three days (Shao et al., 2010) to every ten days (Karuku et al., 2014). Since no irrigation schedule was specified by the case-study owners, a three-day and a ten-day schedule are used to cover a wide range. Furthermore, the most extensively used soil and water management intervention in agriculture is subsurface or tile drainage (TD) (Williams et al., 2015). Tile drains are pipes installed underground to collect percolating irrigation water and enable drainage water recycling (DWR) (Ghirardini and Verlicchi, 2019), which is the practice of collecting drained water from fields in a reservoir for use in times of soil water deficit (Reinhart et al., 2019). Tile drainage collection also helps to reduce nutrient load to water reservoirs by preventing the discharge of nutrient-rich irrigation water (Reinhart et al., 2019). The effect of DWR on the water and nitrogen circularity is analysed. Therefore, the circularity results from the original MCI and the modified approach are compared for four alternatives: (1) Irrigation using freshwater from the river and industrial nitrogen fertilizer application every three days (FW_3); (2) Fertigation using treated wastewater every three days (TW_3); (3) Fertigation using treated wastewater every ten days (TW_10); (4) Fertigation using treated wastewater every ten days with drainage water recycling (TW_10_DWR).

Water and nitrogen supply

To assess the water circularity of this resource recovery solution, the irrigation water quantity is required, which depends on the choice of the crop, the climate, and the irrigation method. For the assessment, a field area of 200,000 m^2 (20 ha) is assumed, and the method of irrigation is drip irrigation with an irrigation efficiency of 85%. An irrigation efficiency (IE) of 100% means that all the irrigation water supplied is used either for a crop's evapotranspiration (ET) or stored in the soil for future use (Malik and Dechmi, 2019). Thus, an 85% IE implies that 15% of the supplied water is neither part of ET nor stored in the soil. This water is assumed to be evaporated during the water application. The tomato crop is irrigated using treated wastewater which requires 400–600 mm of water over its growing season of 90–150 days (FAO, 2022b), which translates to 80,000–120,000 m^3 for a 200,000 m^2 field. The growing period is 108 days, from 1 June 2021 to 16 September 2021. The climate data for the nearest (Palermo) weather station was obtained using the CLIMWAT tool (FAO, 2022a). CLIMWAT is a climate database that enables the calculation of crop water requirements, irrigation supply, and scheduling based on climate data across the globe. Based on the temperature data from the Palermo weather station gathered using CLIMWAT and the crop coefficient obtained from FAO (2022b), shown in Figure 3.3, the ET requirement for tomatoes is estimated over the growing season to be in the order of 112.1×10^3 m^3.

Adding up monthly rainfall data obtained using CLIMWAT, total rainfall in the order of 10.6×10^3 m^3 is estimated, and a net irrigation requirement of 101.5×10^3 m^3 is obtained as the difference between ET requirement and rainfall. Assuming an 85% irrigation efficiency gives 119.4×10^3 m^3 of gross irrigation water estimation, as shown in Table 3.1. The complete calculations are shown in the Supplementary material (S3.1-3.9)

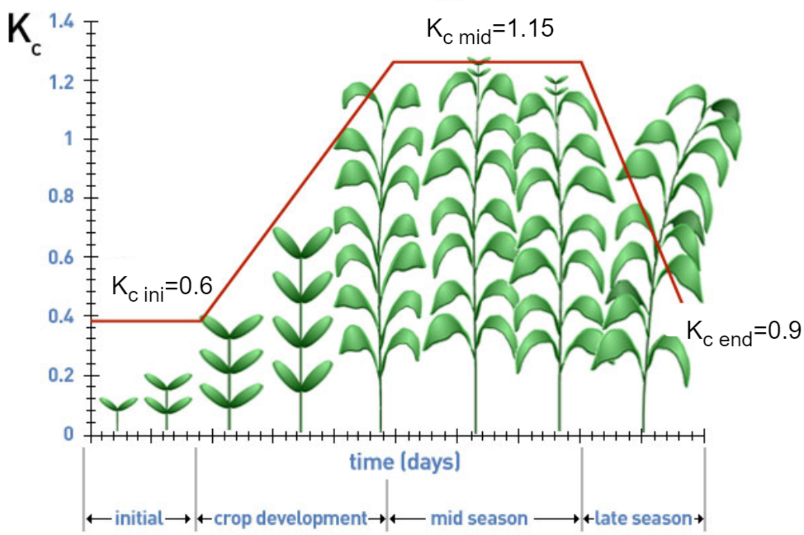

Figure 3.3: Tomato crop evapotranspiration coefficient (Kc) for the different crop development stages (initial, mid, and end). Values obtained and figure adapted from FAO (2022a).

Since TW nitrogen can serve as a secondary source of fertilizers, the nitrogen circularity is also calculated. The total nitrogen concentration in the Corleone effluent is 20 mg N/L (or 0.02 kg N/m^3 water), which lies within the concentration range of 5 and 30 mg N/L specified by Chojnacka et al. (2020). With a total fertigation water requirement of 119.4×10^3 m^3 (see Table 3.1), this means a total of 2.4×10^3 kg nitrogen is applied to the 20 ha field over the growing season. This is equivalent to 120 kg N/ha which falls within the 100 to 150 kg N/ha range of nitrogen requirement for tomato crops as specified by FAO (2022b). Next, the three and ten-day irrigation schedules are entered into the Decision Support System for Agrotechnology Transfer (DSSAT) tool.

DSSAT is a tool developed by an international network of scientists to integrate the knowledge of soil, crops, climate, and management for better decision-making in agriculture (Jones et al., 2003). DSSAT is used to model the fates of irrigation water and the nitrogen used for the fertilization of crops. The soil type is specified as deep sandy loam, and the field size is specified as 200,000 m^2 into DSSAT. The tomato crop and Sunny S–D 2010 cultivar are selected. The initial soil water content on the day of planting (01.06.21) was 106 m^3. Based on the irrigation schedule and the precipitation, DSSAT calculates the soil water content on harvest day. The difference between the soil water on harvest and planting days has been specified as soil water used. In the next section, the obtained circularity results of the original and the modified MCI methods are presented.

Table 3.1: Irrigation water requirement for a 20 ha tomato field in Corleone, Italy, based on ET requirement and rainfall over the entire crop growing season from 01.06.21 to 16.09.21.

Month	**Days**	**ET** ($\times 10^3 m^3$)	**Rainfall** ($\times 10^3 m^3$)	**Net irrigation requirement** ($\times 10^3 m^3$)	**Gross irrigation requirement** ($\times 10^3 m^3$)
June	30	18.2	1.8	16.4	19.3
July	31	40.8	0.6	40.2	47.3
Aug.	31	40.1	3.7	36.4	42.8
Sept.	16	13	4.5	8.5	10
Total	**108**	**112.1**	**10.6**	**101.5**	**119.4**

3.4. Results

3.4.1. Nitrogen and water fates

DSSAT was used for simulating irrigation water and nitrogen fates for the four alternatives to calculate the soil water and nitrogen balance components. The soil water balance is shown in Table 3.2. The nitrogen balance can be found in the Supplementary material S3.6.

Table 3.2: Water inflows and outflows from DSSAT for the irrigation and rainwater specified in our case-study. The water quantities are expressed in m^3/GS where GS stands for the growing season of the tomato crop running from 1 June to 16 Sept.

	FW_3 (x10^3 m^3/GS)	**TW_3** (x10^3 m^3/GS)	**TW_10** (x10^3 m^3/GS)	**TW_10_DWR** (x10^3 m^3/GS)
Inflows				
Treated wastewater input	0	119.4	119.4	119.4
Freshwater input	119.4	0	0	0
Precipitation	10.6	10.6	10.6	10.6
Soil water used	17.1	17.1	13.1	13.1
Outflows				
Irrigation loss	17.9	17.9	17.9	17.9
Groundwater infiltration	6.8	6.8	18.6	6.8
Drainage collected	0	0	0	11.9
Soil Evaporation	89.5	89.5	69.5	69.5
Transpiration	29.5	29.5	33.3	33.3
Crop uptake	3.3	3.3	3.7	3.7

The irrigation water inflow and the precipitation were specified by the authors, and DSSAT estimated the soil water used, which differs for the two irrigation schedules. The soil water used (difference between soil water content on the planting and the harvest day) is higher in the case of TW_3 (17.1×10^3 m^3) than for TW_10 (13.1×10^3 m^3). This may be due to the crop using more soil water in the three-day interval case due to higher evaporative losses. Further, the groundwater infiltration is higher for the TW_10 (18.6×10^3 m^3) than for the TW_3 (6.8×10^3 m^3) case. In the TW_10_DWR case, part of the drainage water (11.9×10^3 m^3) is collected for reuse, while in the rest of the cases, all of the drainage water is assumed to recharge groundwater. While the soil evaporation is higher for the three-day interval (89.5×10^3 m^3) than the ten-day interval irrigation (69.5×10^3 m^3), the transpiration and crop uptake are higher for the ten-day interval case (3.7×10^3 m^3).

The industrial and wastewater nitrogen input was specified, and DSSAT calculated the soil nitrogen used and mineralized nitrogen. Soil nitrogen used for the TW_10 and TW_10_DWR alternatives was higher (0.3×10^3 kg N/GS) than for the TW_3 alternative (0.2×10^3 kg N/GS). This may be because, in the lower frequency applications, the crop relies more heavily on internal nitrogen cycling (Dawson et al., 2008). Nitrogen loss with drainage was higher for the TW_10 case (0.7×10^3 kg N/GS) than for the TW_3 case (0.4×10^3 kg N/GS) because more water infiltrates with lower frequency fertigation, also draining the nitrogen along with it. TW_3 had a higher crop uptake of nitrogen (3.6×10^3 kg N/GS) as compared to the TW_10 and TW_10_DWR alternatives (3.5×10^3 kg N/GS) because high-frequency fertigation leads to higher crop uptake of nitrogen (Farneselli et al., 2015).

3.4.2. Circularity assessment

The original MCI method was applied to the above case-study. For the FW_3 alternative, both the water and nitrogen circularity values are 0. This is because freshwater was used for irrigation along with industrial nitrogen fertilizer, and on the output side, none of the nitrogen or water flows can be considered circular. The water and nitrogen circularities for the TW_3 alternative are 41% and 29%, respectively.

This improvement (relative to FW_3) is due to the use of treated wastewater containing nitrogen. For the TW_10 case, water circularity improves further to 42%. This slight improvement results from lower net soil water used in the ten-day interval than in the three-day interval irrigation schedule. The nitrogen circularity remains at 29% for TW_10. Finally, the water circularity of 46% and nitrogen circularity of 37% is obtained in the TW_10_DWR alternative. The higher water and nitrogen circularity values obtained in this alternative (compared to the TW_10 alternative) are due to the drainage water collected for reuse.

The new MCI method was applied to the same case-study. The water and nitrogen flows were classified as linear, restorative or regenerative flow. Figure 3.4 shows the difference between regenerative and linear water flows as part of the original and the modified MCI. Precipitation, FW irrigation, and soil water use do not originate from reuse/recycle sources and, thus, are linear flows. Evaporation and transpiration are linear flows because the water used in liquid form is returned to the atmosphere as vapour. It was assumed that the water contained in the tomato would re-enter the WWTP through the human diet. Also, it was assumed that the crop residue is mulched and its water content replenishes soil water. Therefore, the crop uptake of water was treated as restorative. Irrigation water infiltration can contribute to groundwater recharge (Jia et al., 2020), and thus, infiltration was considered to be a regenerative flow.

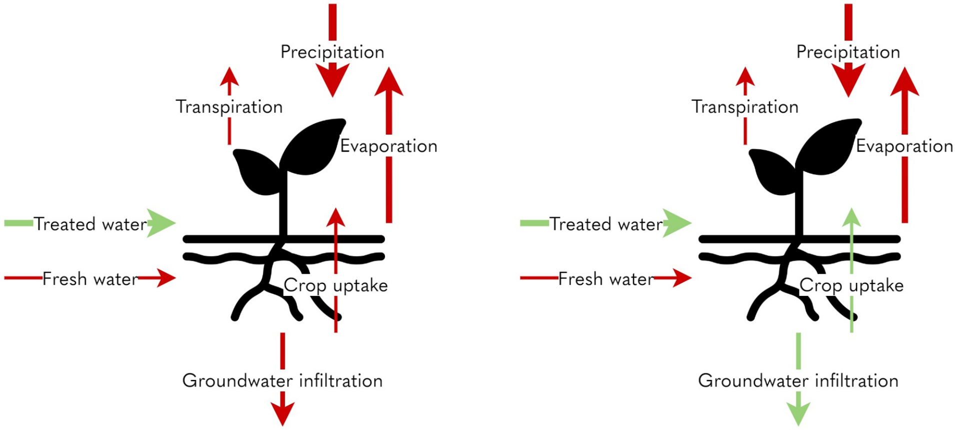

Figure 3.4: (a) Restorative/regenerative (green) and linear (red) water flows based on the original MCI. (b) Regenerative (green) and linear (red) flows based on the modified MCI. In the modified approach, flows such as crop uptake and groundwater infiltration of water are counted as restorative/regenerative flows unlike in the original MCI approach.

Regarding nitrogen flows, soil nitrogen use, industrial nitrogen fertilizer addition, and soil nitrogen mineralization are considered linear input flows since they do not originate from any reuse/recycle sources. But, nitrogen added with treated wastewater is a restorative flow. Further, losses in the form of ammonia (NH_3) or nitrogen oxide (NO) are both linear output flows because they are not reused/recycled. However, N_2 loss is a form of regenerative flow because nitrogen is released in the form in which it was originally obtained from the atmosphere. To illustrate the new MCI method, the water MCI calculation steps for the TW_3 alternative are shown here.

1. Calculate total water and nitrogen inflows as follows:

$$
\begin{aligned}
M_{\text{water}} &= \text{Gross irrigation water} + \text{Precipitation} + \text{Reduction in soil water} \\
&= (119.4 + 10.6 + 17.1) \times 10^3 = 147 \times 10^3 m^3
\end{aligned}
\tag{3.7}
$$

$$
\begin{aligned}
M_{\text{N}} &= \text{Fertigation N} + \text{Reduction in soil N} + \text{Mineralized N} \\
&= (2.4 + 0.2 + 1.5) \times 10^3 = 4.1 \times 10^3 m^3
\end{aligned}
\tag{3.8}
$$

2. Calculate virgin inflows for water and nitrogen as follows:

$$V_{\text{water}} = 147(1 - \frac{119.4}{147.1}) \times 10^3 = 27.7 \times 10^3 m^3 \tag{3.9}$$

$$V_{\text{N}} = 4.1(1 - \frac{2.4}{4.1}) \times 10^3 = 1.7 \times 10^3 kg \tag{3.10}$$

3. Calculate unrecovered water and nitrogen outflows as follows:

$$W_{\text{water}} = 147(1 - \frac{6.8}{147.1} - \frac{3.3}{147.1}) \times 10^3 = 137 \times 10^3 m^3 \tag{3.11}$$

$$W_{\text{N}} = 4.1(1 - \frac{3.6}{4.1} - \frac{0}{4.1}) \times 10^3 = 0.5 \times 10^3 kg \tag{3.12}$$

4. Calculate the linear flow indicators as follows:

$$LFI_{\text{water}} = \frac{27.7 + 137}{2 \times 147.1} = 0.56 \tag{3.13}$$

$$LFI_N = \frac{1.7 + 0.5}{2 \times 4.1} = 0.27 \tag{3.14}$$

5. Calculate the material circularity indicators as follows:

$$MCI_{\text{water}} = (1 - 0.56) \times 100 = 44\% \tag{3.15}$$

$$MCI_N = (1 - 0.27) \times 100 = 73\% \tag{3.16}$$

Table 3.3 shows the results of the modified MCI and the original MCI and Table 3.4 shows the averages of the water and nitrogen circularities for each alternative. As can be seen, when compared to the circularity assessed using the original MCI method, the modified MCI method shows higher water and nitrogen circularity values for all four alternatives. This higher circularity is because of the consideration that both the water taken up by the crops and the water infiltrating underground contribute to the circular economy. Groundwater infiltration of the irrigation water may increase the cost for the farmers. However, it is known that improving the water use efficiency of irrigation often comes at the cost of groundwater recharge (Ebrahimi et al., 2016; Jiménez-Martínez et al., 2009; Xu et al., 2010). So, from the perspective of resource availability alone (despite the increased economic costs for the farmers), groundwater recharge does contribute to water circularity. A higher nitrogen circularity is because of the consideration that the plant uptake of nitrogen is restorative and N_2 emission is a regenerative flow. The average values show that TW_10_DWR is the overall best performing alternative with an average circularity of 41% using the original MCI and 64% using the modified MCI.

As seen in Table 3.3, water circularity improved from 3% to 44% because of switching to reused water. Nitrogen circularity improved from 41%

Table 3.3: The circularity of the alternatives using the original and the modified MCI methods. FW_3: Freshwater irrigation+industrial fertilizer; TW_3: Fertigation every three days; TW_10: Fertigation every ten days; TW_10_DWR: Fertigation+drainage recycling.

	Original MCI results (%)		**Modified MCI results (%)**	
Alternatives	Water circularity	Nitrogen circularity	Water circularity	Nitrogen circularity
FW_3	0	0	3	41
TW_3	41	29	44	73
TW_10	42	29	50	71
TW_10_DWR	46	37	50	78

Table 3.4: The averages of the water and nitrogen circularities for the four alternatives. FW_3: Freshwater irrigation+industrial fertilizer; TW_3: Fertigation every three days; TW_10: Fertigation every ten days; TW_10_DWR: Fertigation+drainage recycling.

	Original MCI results (%)	**Modified MCI results (%)**
Alternatives	Average circularity	Average circularity
FW_3	0	22
TW_3	35	59
TW_10	35	60
TW_10_DWR	41	64

to 73% when replacing industrial fertilizers with TW-N. A lower fertigation frequency improved water circularity from 44% to 50% while reducing nitrogen circularity from 73% to 71%. The water circularity improvement is due to lower evaporation losses, and the reduced nitrogen circularity is due to a lower crop nitrogen uptake. Lastly, collecting drainage water for reuse along with the runoff nitrogen improved nitrogen circularity from 71% to 78% because this prevented the dissipation of the TW-N. However, this intervention did not affect the water circularity because it was assumed that the uncollected drainage contributes to groundwater recharge.

The nitrogen and water circularities for the alternatives are shown in Figure 3.5. Switching from FW and industrial nitrogen fertilizers to TW fertigation leads to the largest circularity improvement. Reducing fertigation frequency improves water circularity but reduces nitrogen circularity. However, the decrease in nitrogen circularity can be offset if the nitrogen in the drainage water can be collected for reuse. The complete results can be found in the Supplementary material S3.4, S3.5, S3.7, and S3.8.

3.5. Discussion

When comparing the original MCI with the new approach, the circularity of the fertigation case-study is lower using the original MCI method (e.g., 42%

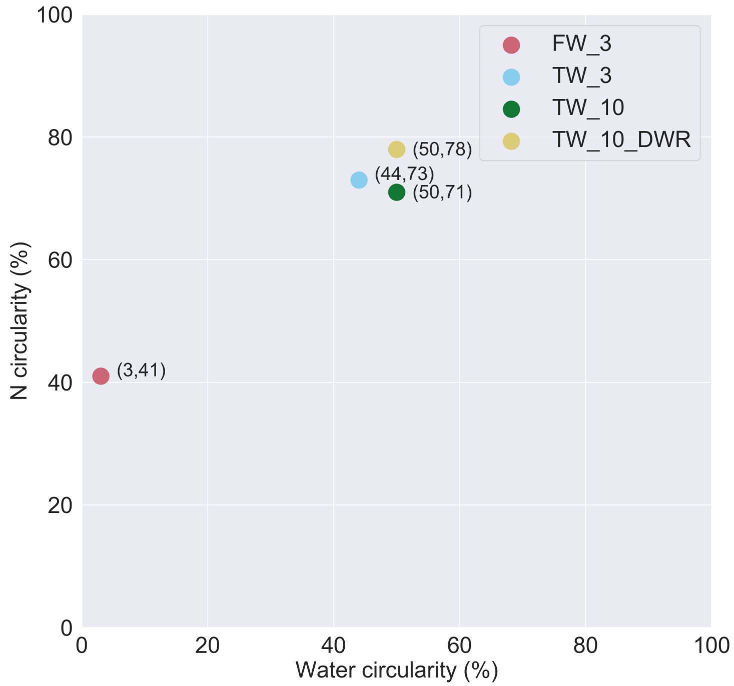

Figure 3.5: Water and N circularities for the different fertigation schedules and drainage management alternatives using the modified MCI method.

water circularity for TW_10 using the original MCI compared to 50% using the modified MCI). In the original method, only recycling/reuse of resources for human functions can be considered circular since the original MCI and most other methods were developed for the technical cycle resources. This is remedied by the modified approach to the MCI assessment.

The first factor leading to a higher water circularity of the modified MCI is the consideration of groundwater infiltration as a regenerative flow. Treated wastewater fertigation is mostly practised in arid and semi-arid regions of the world (Farhadkhani et al., 2018; Elgallal et al., 2016). Additionally, irrigation water is a major source of groundwater recharge flows, especially in arid and semi-arid regions (Jiménez-Martínez et al., 2009; Qin et al., 2011), contributing to future water availability. Thus, groundwater infiltration should be considered a contribution to circularity, even though it is an irrigation loss. According to the definitions provided in this paper, the groundwater infiltration flow falls under a regenerative type flow.

A disadvantage of the new method is that it does not account for the quality of the infiltration water. Reuse of treated wastewater for fertigation can lead to excess biogenic compounds and pharmaceuticals leaching into groundwater (Chojnacka et al., 2020). The impact of water quality should also be accounted for when considering the holistic sustainability impact of fertigation, but the method only deals with the quantity of water since the focus is on the circularity aspect alone.

The second factor is crop water uptake. The total crop water uptake comprises the water in the edible as well as the non-edible parts. Here, it was assumed that the water flowing into the edible part is a restorative flow because this water will be directly used for human consumption. Further, it was assumed that the rest of the water would remain in the soil because the non-edible parts of the crop could be cut and left on the soil. Both of these are simplifications as some water will evaporate, and more accurate models are required to quantify such losses. For now, the total plant uptake of water is considered to be a restorative flow.

Similarly, the nitrogen circularities from the modified MCI are higher than those from the original MCI (e.g., 29% for TW_3 by the original MCI and 73% by the modified MCI). The reason for this is the crop uptake of nitrogen. The nitrogen uptake to the edible part of the crop is a restorative flow because this nitrogen is meant for direct human use. A large quantity of nitrogen taken up by the non-edible part of a crop mostly remains in the soil after harvest (Fan et al., 2014; Gao et al., 2022, 2020) and hence may be available for the subsequent crop (Poudel et al., 2001). Enhancing plant uptake of nitrogen is necessary for sustainability because this can prevent nitrogen dissipation (Chen et al., 2019; Dimkpa et al., 2020).

It may be a simplification to assume that all of the nitrogen uptake remains in the soil after harvesting. Using more accurate nitrogen fate models is recommended to estimate exactly how much of the plant uptake nitrogen remains in the soil after harvest, but, for now, the plant uptake of nitrogen can be considered a restorative flow.

With the modifications to the MCI method, the circularity assessment was aligned better with sustainability in two ways. Firstly, by defining regeneration as the return of resources to the state in which the resources were appropriated from the natural environment, it was ensured that flows such as groundwater infiltration and N_2 emissions count towards circularity improvement even though they are fertigation losses. Furthermore, crop uptake of water and nitrogen was considered to be restorative flows which means that maximising these flows also translates into improved circularity. This is logical because increasing the crop uptake of these resources can improve agricultural productivity and reduce losses.

Secondly, the circularity assessment was based on a resource flow model. Recycling a biogeochemical resource does not necessarily lead to a high circularity because of potential environmental losses such as the evaporation of reused water. A drastic reduction in river flows due to the growing consumptive use through evapotranspiration is a well-known phenomenon (Falkenmark and Lannerstad, 2005; Zisopoulou and Panagoulia, 2021). Without estimating environmental losses, which depend on local conditions, one risks an inaccurate assessment. It was shown how factors such as fertigation schedule and drainage water management could affect the circularity results.

Although the new approach was demonstrated on the short-timescale biogeochemical resources involved in treated wastewater fertigation, the approach can be applied to any resource recovery solution related to the water sector that deals with biogeochemical resources. This is because no restrictions were introduced by the modification presented in this study; rather, only the scope of application of the MCI method was extended by introducing some details related to the short-term biogeochemical cycles.

Coming to the case-study specific discussion, overall the option of using treated wastewater for fertigation along with drainage water reycling led to the maximum circularity of 64%. The circularity values of fertigation are affected by the water and nitrogen fates which in turn are affected by the fertigation schedule and possibly other factors such as climate and rainfall. However, it was found that changing the fertigation frequency has opposite effects on water and nitrogen circularity values. The water circularity was found to be 50% when the field was irrigated every ten days as compared to 44% for every three days irrigation. Evaporative losses increase with a higher irrigation frequency (Mermoud et al., 2005; Mukherjee et al., 2010) since a smaller depth of water applied with a higher frequency leads to superficial wetting of soil, causing high evaporation (Mermoud et al., 2005) and lower infiltration, thus reducing circularity.

Interestingly, the opposite effect of fertigation frequency is observed for nitrogen circularity which decreases from 73% to 71% when shifting from a three-day to a ten-day interval. This effect may be because crop uptake of nitrogen is known to increase with higher frequency fertigation of the tomato crop (Farneselli et al., 2015). The opposite effects on circularity based on the fertigation frequency means that careful planning is necessary to maximize both the circularities of water and nitrogen. This also means that in some cases, recovering nitrogen from wastewater and using it separately from irrigation water may be advisable for optimal circularity.

Collecting drainage water for reuse did not show any effect on the water circularity, which remained at 50%. This is because water drained from the irrigated field was considered to be contributing to circularity regardless of being collected. In arid and semi-arid regions, where fertigation is most practised, infiltrating irrigation water is one of the major groundwater recharge flows (Ebrahimi et al., 2016; Jiménez-Martínez et al., 2009; Xu et al., 2010) and therefore, this flow was considered to be regenerative. If part or whole of the infiltrating water is collected for reuse using controlled drainage methods, then the collected flow will count as a restorative flow. In either case, the same water circularity is achieved. On the contrary, nitrogen circularity improved from 71% to 78% for the TW_10_DWR alternative. Collecting drainage water for reuse is known to reduce nitrogen loss (Reinhart et al., 2019; Williams et al., 2015) and thus contributes to improved nitrogen circularity, as confirmed by the assessment.

3.6. Conclusions

This study aimed to develop a new and improved circularity assessment approach for resource recovery solutions in the water sector. The following conclusions can be drawn about the novel approach:

- The novel method resulted in higher circularity values compared to the original MCI method. This was because the new approach accounts for the fact that certain biogeochemical flows commonly classified as losses can contribute towards circularity.
- The new definitions of restorative, restorative, and regenerative flows will help to classify common resource flows for an accurate circularity assessment.

The case-study results provide the following conclusions:

- Overall, the option of using treated wastewater with drainage water recycling leads to the maximum circularity for the case-study.
- Water circularity can be significantly improved with treated wastewater fertigation, especially with a low-frequency schedule. However, irrigation also leads to substantial water losses, mainly in the form of evapotranspiration and infiltration. While some of these losses (e.g., evapotranspiration) reduce water circularity, other losses (e.g., groundwater infiltration) may even contribute to water circularity.
- Substituting industrial fertilizers with the nutrients contained in treated wastewater improves nitrogen circularity. However, contrary to water circularity, a lower application frequency can decrease nitrogen circularity by reducing its crop uptake. Nitrogen lost with infiltrating water should be collected for reuse, to improve nitrogen circularity.
- A high-frequency application of recovered nitrogen and a low-frequency application of treated wastewater along with drainage water collection will improve the overall circularity.

To conclude, the new approach to circularity assessment works well and can help to optimize the resource recovery solutions in the water sector. Although more accurate resource flow models and further discussion on restorative and regenerative flows are needed, the new approach is a crucial step towards ensuring a more circular and sustainable water sector.

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4

Assessing the benefits of the recovered resources to nature

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4.1. Introduction

Municipal WWTPs protect the natural environment and humans from the discharge of untreated domestic wastewater which is a health hazard (Mo and Zhang, 2013; Van Der Hoek et al., 2016). However, wastewater is also a source of valuable resources such as nutrients and organic matter which may be recovered to replace virgin resources (Chrispim et al., 2020).

Resource recovery can reduce the negative environmental impacts of WWTPs such as carbon footprint and eutrophication (Cornejo et al., 2016). This can result from lower energy use within WWTPs or the avoided burden of extracting virgin resources. Furthermore, recovery of resources can improve the economic efficiency of WWTPs either by reducing treatment costs or generating extra revenue. However, resource recovery may not only reduce the environmental and economic costs but can also actively benefit the natural environment.

Using the recovered resources, human society can provide a reciprocal service to the natural environment in return for the resources that nature provides. Assessing the potential nature benefits will help foster a symbiotic relationship between human society and the natural environment. This can also create a more holistic view of wastewater treatment and thereby reveal more resource recovery opportunities (Trimmer et al., 2019). Furthermore, it is not sufficient to merely think of damage reduction, we need to explore the potential for actively benefiting the natural environment wherever possible (Bhambhani et al., 2022). However, the assessment of this nature-benefiting aspect of resource recovery from wastewater has largely remained neglected (Trimmer et al., 2019).

The services provided by the natural environment to human society are often considered a one-way flow of benefits and Comberti et al. (2015) point to the need to reconsider this idea. The same authors mention the need to introduce 'reciprocal benefits' (i.e., from humans to nature) within the concept of sustainability. Moreover, two human viewpoints can be distinguished: one in which humans are a part of nature and another wherein humans are considered to be autonomous entities that rule over nature. The former is a viewpoint that more people find congruous with their experiences (Jax et al., 2018). Yet most common sustainability discourses are based on a unidirectional flow of benefits from nature to humans and methods backed by a reciprocal view must be developed (Jax et al., 2018). More specifically, in the wastewater treatment sector, Trimmer et al. (2019) contributed a conceptual framework that explains the potential of the recovered resources from WWTPs to enhance the ecosystem services. However, the framework does not provide a method to quantitatively assess the enhancement and Trimmer et al. (2019) suggest the development of an assessment method for future work, resulting in a corresponding research gap that is addressed in this chapter.

Therefore, the objective of this chapter is to develop a novel method to assess the potential nature benefits of the resources recovered from municipal wastewater. This study focuses on certain key resources that include water, nutrients, and organic matter. This is not meant to replace the eco-efficiency assessment methods but to complement them as also suggested by Jax et al. (2018). This chapter presents nature benefits as the next step towards sustainability.

In Section 4.2, the main resources present in domestic wastewater are introduced. This is followed by a discussion on the potential benefits of resource recovery including improved economic and eco-efficiencies. Further, the potential nature benefits from the resources recovered from wastewater and the importance of assessing these benefits are discussed. In Section 4.3, the novel method for quantitatively assessing the nature benefits is explained and a real-life case-study is described. Section 4.4 presents the results of the case-study, including the uncertainty and sensitivity analyses. In Section 4.5, the wider implications of the assessment are discussed along with the limitations of the method. Finally, the conclusions about the method and its application are presented in Section 4.6. The calculations are shown in Supplementary material (S4.1-4.7).

4.2. Background

4.2.1. Resources present in domestic wastewater

The focus of WWTPs has traditionally been the removal of pollutants from sewage, but now includes resource recovery (Renfrew et al., 2022; Van Der Hoek et al., 2016; Wang et al., 2015; Zhang et al., 2024). This is necessary because important resources are becoming scarce with increasing human population (Van Der Hoek et al., 2016).

Several resource recovery pathways have been studied but a few of them gather the most attention. Mo and Zhang (2013) discussed the three main pathways for resource recovery, namely water reuse, on-site energy generation, and nutrient recycling. Trimmer et al. (2019) listed the three most common categories of resources to be recovered from wastewater: water for reuse, nutrients, and organic matter. Energy recovery, organic carbon (C), and nutrient recovery were also discussed by Puchongkawarin et al. (2015). Kehrein et al. (2020) discussed the potential of recovering P as struvite and organic (expressed as COD, chemical oxygen demand) as energy or bio-polymers.

The upper limit of the quantity of waste generation or resource exploitation that earth can sustain can be summarized by the nine planetary boundaries of Rockström et al. (2009). Four out of the nine planetary boundaries (species extinction rate, atmospheric CO_2 concentration, and the emissions of reactive N and P) are transgressed already. Three of the four transgressed boundaries can be traced to the mismanagement of , N, and P (Slootweg, 2020). The recovery of nutrients and organic matter is urgently needed to prevent further emissions of C, N, and P. Furthermore, organic matter, nutrients, and water are the most valuable resources that can be recovered from WWTPs (Lee et al., 2013; Mo and Zhang, 2013; Verstraete et al., 2009) and data regarding the mass balances of C, N, and P are relatively easy to estimate (Nowak et al., 1999).

Given the above, the resources most commonly recovered from WWTPs include treated wastewater, organic matter, and nutrients (mainly N and P). Hence, they will be the focus of this chapter.

4.2.2. Benefits of resource recovery

Numerous benefits of recovering resources from a WWTP have been discussed including economic value generation, resource circularity, reduced eutrophication, reduced ecotoxicity, improved energy efficiency and carbon footprint offset (Coma et al., 2017; Gherghel et al., 2019; Kehrein et al., 2020; Lam et al., 2022; Ruiken et al., 2013). These benefits can be classified broadly under two categories: improved eco-efficiency and enhanced economic efficiency of a WWTP. Methods to quantify these benefits have also been developed and continue to be the focus of studies.

Eco-efficiency is the ratio between the service delivered by a process and the negative environmental impacts of the process (Hauschild and Huijbregts, 2015). Therefore, the eco-efficiency of a WWTP can be defined as the ratio between the volume of wastewater (m^3/y) treated to discharge standards and the environmental impacts of the treatment process (e.g., the climate change impact measured in kg CO_2 eq.). Most conventional WWTPs have a net negative environmental impact due to their high resource use intensity (Hao et al., 2019; Schaubroeck et al., 2015). However, resource recovery can reduce the negative environmental impacts of WWTPs (Cornejo et al., 2013; Hao et al., 2019) thereby, improving their eco-efficiency. E.g., Cornejo et al. (2013) discussed the reduced eutrophication potential of a WWTP because of treated wastewater (TW) reuse for fertigation (application of fertilizers via irrigation), reduction in the carbon footprint, and embodied energy resulting from energy recovery.

The resources recovered from WWTPs generally have a higher cost than the virgin resources they replace. However, the higher cost can be offset by the reduced operational costs of the WWTP. E.g., in the Amsterdam West WWTP, an investment cost of € 4 million was estimated for struvite recovery which can result in an expected yearly saving of about € 400,000 for maintenance (van der Hoek et al., 2017). WWTPs can also generate revenue by selling the recovered resources thus, generating extra revenue (Tarpani and Azapagic, 2018). Therefore, resource recovery can improve the economic efficiency of a WWTP by reducing the operational costs of the water treatment or generating revenue from the sale of the recovered resources.

4.2.3. Nature benefits from resource recovery

Along with a lower negative environmental impact, positive effects on the natural environment can also be achieved using the recovered resources. Soil fertility, microbial biomass, and soil enzyme activity can be improved using sewage sludge application (Boudjabi and Chenchouni, 2021; Dhanker et al., 2021). TW discharge can help with improvement in surface water quality, bank stabilization, and the return of pollution-sensitive aquatic species (Bischel et al., 2013).

A method to assess these potential benefits is necessary. First, this will support a cycle of reciprocal benefits between human society and nature (Trimmer et al., 2019). The natural environment provides numerous services to human society that can be conceptualized using the ecosystem services framework (Wallace, 2007). Human society can also potentially provide benefits to the natural environment. An example of this is the indigenous communities enhancing the soil fertility of the Amazon forests by adding charcoal, bones, and manure (Comberti et al., 2015). Mutually beneficial relationships (symbiosis) among organisms and between organisms and their natural environments are common. Yet, human society's consideration of their beneficial role in nature has remained limited. The research focus remains on the reduction of negative impacts. The opportunity for benefiting nature that resource recovery provides us should be explored further.

Second, assessing the potential nature benefits will lead to a more holistic view of wastewater treatment and thereby may reveal more resource recovery opportunities (Trimmer et al., 2019). To illustrate, adding organic matter can improve soil structure and reduce erosion with secondary benefits such as improved water retention and enhanced vegetative growth. In some contexts, these benefits can be highly valuable and should not be ignored in favour of a directly recognizable benefit to humans, such as energy generation (Trimmer et al., 2019).

Third, only damage reduction is insufficient for sustainability, especially for the emissions that have crossed sustainable planetary limits (Bhambhani et al., 2022). For example, the anthropogenic emissions of reactive N and P have crossed the limits that planet earth can sustain (Sandström et al., 2023; Steffen et al., 2015). In such cases, an active approach towards repairing the nutrient flows is required. Despite the knowledge of potential nature reciprocity and the need to assess it, methods incorporating the assessment of enhanced ecosystems are rare (Trimmer et al., 2019) and usually qualitative. Therefore, a method for the quantitative assessment of natural environmental benefits is needed.

The concept of actively providing benefits to the natural environment has to be defined first. In this chapter, the natural environment is represented by stock and flux models of water, nutrients (N, P), and carbon. The resources move between these stocks by natural or artificial processes e.g., N present in the atmosphere flows to the soil stock through natural fixation. It is not the intention to comprehensively describe the natural environment using a limited number of elements but to focus our attention on the most relevant parts of nature that a WWTP can affect. Next, environmental damage is conceptualized as an excessive build-up of a resource in a particular stock or an excessive removal of a resource from a stock. E.g., whereas climate change can be looked upon as an excessive build-up of carbon in the atmospheric stock (Ajani et al., 2013), water scarcity can be conceptualized as an excessive removal of water from a groundwater stock.

Nature reciprocity can be defined as a re-balancing of the resource stocks. This can be achieved by directing a resource from one stock to another that could benefit from it e.g., excessive use of fertilizers has caused a build-up of reactive nitrogen (N_r) species in European rivers (Blaas and Kroeze, 2016). Future N_r emissions can be redirected to living biomass with the help of WWTPs. The authors do not suggest that pristine natural conditions can be reached again but the focus is on including the assessment of nature benefits as the next step in the sustainability pursuit.

4.3. Methods

The focus of this chapter is on three resource categories and three pathways through which they can benefit nature. The resources are treated wastewater (TW), nutrients (N and P), and organic matter (OM). The TW can be used to restore freshwater; the recovered nutrients can be used for nutrient cycling through the pathways of biomass assimilation; and the recovered organics can support carbon cycling through soil organic matter (SOM) addition. The links between the recoverable resources and their potential nature benefits are shown in Figure 4.1 and are based on the work of Trimmer et al. (2019). As shown in the subsequent sections, the key novelty here is the development of a new method to assess the potential nature benefits of wastewater-recovered resources. This is the first step in this direction and the indicators are kept simple to capture sufficient details but maintain ease of calculation for the decision-makers.

4.3.1. Water cycling through freshwater restoration

By 2050, between a third to a half of the global population is likely to face water scarcity (Boretti and Rosa, 2019; He et al., 2021) mainly driven by growing demand, a reduction in water resources, and pollution (Boretti and Rosa, 2019). The global withdrawal of blue water (groundwater and surface water) should remain under 4000-6000 km^3/y to avoid the irreversible collapse of ecosystems (Rockström et al., 2009). Thus, preserving and restoring freshwater reservoirs is critical and WWTPs can play a crucial role here.

As the quality of the TW improves with technology, WWTPs can be seen as significant contributors to freshwater reservoirs (Verstraete et al., 2009; Wang et al., 2017). Stream flow augmentation using TW discharge can restore freshwater and improve habitats for aquatic ecosystems (Plumlee et al., 2012). However, the quality of the TW is crucial to the restoration of the freshwater reservoirs. Assuming the wastewater treatment is achieved to meet the discharge standards, the pollutants present still re-

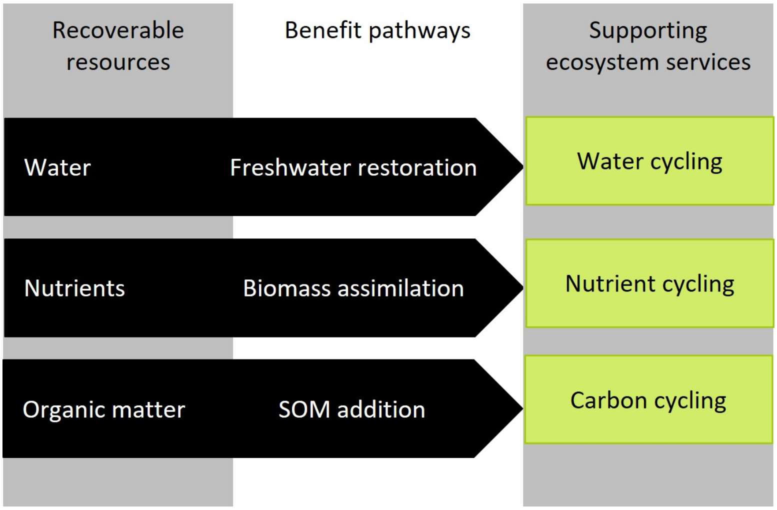

Figure 4.1: The link between resources recovered from domestic wastewater and the potential positive effects on the natural environment through enhancement of supporting ecosystem services, based on Trimmer et al. (2019).

quire a certain quantity of freshwater to be diluted to background concentrations. Therefore, the nature benefit is here defined by the quantity of the discharged water multiplied by a factor that accounts for the water quality.

The concepts of grey water footprint (GWF) and water pollution level (WPL) developed by Hoekstra et al. (2011) are used here. The GWF refers to the volume of freshwater required to dilute a given pollutant concentration to the background concentration in the stream (Mekonnen and Hoekstra, 2015) and is calculated as follows:

$$GWF_i = \frac{L_i}{(C_{max} - C_{nat})} \tag{4.1}$$

where GWF_i is the grey water footprint of the WWTP discharge stream in month i, L_i is the pollutant load (kg/month) for month i, C_{max} is the maximum acceptable concentration (kg/m^3) of a pollutant in a stream obtained from the EU water directive (EC, 2000), and C_{nat} is the natural background concentration of the pollutant in the receiving stream when there was no human disturbance in the catchment. If C_{nat} is not known, Hoekstra et al. (2011) suggest using 0 kg/m^3.

A few special conditions should be discussed. It is assumed that in most cases, the C_{max} of a pollutant will be greater than C_{nat} and then Equation 4.1 applies. In the case of C_{max} being equal to C_{nat}, the GWF value would be undefined. However, this situation is unlikely to occur because maximum concentration standards are usually not set equal to the natural background concentrations (Hoekstra et al., 2011). Also, it is unlikely that the C_{max} of a pollutant is specified to be lower than the C_{nat}. Still, if that happens then Equation 4.1 should not be used.

The WPL is the ratio between the GWF of a WWTP discharge stream and the stream runoff (m^3/y) (Mekonnen and Hoekstra, 2015) and is calculated as follows:

$$WPL_{\mathrm{i}} = \frac{GWF_{\mathrm{i}}}{R_{act\,i}} \tag{4.2}$$

where GWF_i is the grey water footprint of the WWTP discharge calculated using Equation 4.1 in the month i, and R_{act_i} is the actual discharge of the stream receiving the TW (m^3/month) in the month i. A smaller WPL value means a better discharge quality. The WPL calculation has to account for the seasonal stream discharge variations and a monthly estimation is enough for this (Hoekstra et al., 2011).

The benefit indicator is here defined as the quantity of TW discharged into nature multiplied by a quality factor, resulting in freshwater restoration (FR) as follows:

$$FR = \sum_{i=1}^{12} Q_{dis_i} \times (1 - WPL_i) \tag{4.3}$$

where FR is the freshwater restored in m^3/y, Q_{dis_i} is the volume of treated water discharge in m^3/month for month i, and WPL_i is the water pollution level calculated using Equation 4.2.

A negative FR implies that there is a net consumption of freshwater for the dilution of the pollutants in the TW. Two examples demonstrate the effect of the WPL on the FR values. The WPL of TW can vary significantly and values >1 and as low as 0.08 have been reported in Wang et al. (2020) describing the status of N discharge from WWTPs in Shenzhen, China. Suppose WWTP1 and WWTP2 each discharge 100,000 m^3/month to a river. The N WPL for WWTP1 and WWTP2 are 1.2 and 0.08. The FR achieved by them are shown below.

FR_{WWTP1} = 100,000×(1-1.2) = -20,000 m^3/month
FR_{WWTP2} = 100,000×(1-0.08) = 92,000 m^3/month

Therefore, the WWTP1 discharge requires more water for dilution leading to a net decrease of freshwater by 20,000 m^3/month. In contrast, WWTP2 with a low WPL restores 92,000 m^3/month of freshwater.

4.3.2. Nutrient cycling through biomass assimilation

Recovering N and P from wastewater is important for three main reasons. First, N and P are crucial for crop fertilization and the production of these fertilizers is energy and resource intensive. A 4% annual increase in fertilizer production is projected until 2050 to feed a growing human population

Xie et al. (2016). The Haber Bosch (HB) process used to obtain reactive N from the atmosphere to manufacture fertilizers consumes 1-2% of the global energy expenditure (Houlton et al., 2019). Contrary to N, P is a non-renewable resource obtained from mining phosphate rocks (van der Hoek et al., 2018). Given the non-renewability of these rocks, P was designated a critical raw material by the EU in 2014 (Hukari et al., 2016). The high energy consumption of the HB process and the excessive mining of the phosphate rocks can be avoided by nutrient recovery from wastewater.

Second, recovering nutrients can prevent eutrophication from TW discharge (Babcock-Jackson et al., 2023; Singh et al., 2023). The loss of reactive N through TW discharge causes direct human health damage such as asthma and cancer and disrupts ecosystems inducing a loss of biodiversity and ecosystem services (Bodirsky et al., 2014). The economic costs of N_r loss to the environment have been estimated to be between €75 and €485 billion in the EU (Van Grinsven et al., 2013) and between $80 and $441 billion in the US (Sobota et al., 2015). P in TW discharge is another major contributor to eutrophication, fish death and ecosystem destruction (Patyal et al., 2022), and causes human health issues such as metabolic bone disease (Gao et al., 2020).

Third, nutrient products derived from wastewater usually have a higher nutrient uptake efficiency (NUE) than conventional fertilizers (Babcock-Jackson et al., 2023; Saliu and Oladoja, 2021; Santos and Pires, 2018). About 85% of the Nr created using the HB process and 90% of the mined P are used for food production (Galloway et al., 2003; Kanter and Brownlie, 2019). However, only between 20% and 30% of the N in the fertilizers is taken up by crops with the rest leaching into groundwater, volatilizing as ammonia, or running off in streams (Naz and Sulaiman, 2016). Similarly, most of the P is lost to the environment since the crop uptake of P is known to be under 25% (Roberts and Johnston, 2015). One of the ways to deal with the low NUE is by using slow-release fertilizers (Babcock-Jackson et al., 2023). The industrial manufacturing of slow-release fertilizers is limited by the high production cost and the need for petroleum-based polymers (Vejan et al., 2021). Here, wastewater-recovered nutrients offer an advantage because these are usually in an adsorbed or encapsulated form ensuring their slow release (Vejan et al., 2021).

N is active in the natural environment until either sequestered or converted to N_2 (Galloway et al., 2021). Since biomass assimilation can combat excess reactive nutrient species in the environment (Xu and Shen, 2011) and the nutrient products derived from wastewater tend to have a higher NUE, the WWTPs can provide a nature benefit by efficiently assimilating nutrients into plant biomass.

The WWTP nutrient recovery efficiency (NRE) can vary between the different recovery techniques (Xie et al., 2016). Also, different nutrient recovery products have varying NUE (Sigurnjak et al., 2016). Thus, both these factors will impact nutrient assimilation. An equation to measure the biomass assimilation of nutrients is presented below:

$$BA = M_{\text{inf}} \times NRE \times NUE \tag{4.4}$$

where BA is the biomass assimilation of nutrients in kg/y, M_{inf} is the mass of nutrients entering a WWTP with the influent in kg/y, NRE is the nutrient recovery efficiency (0-100%), NUE is the nutrient uptake efficiency (0-100%).

Suppose two WWTPs are compared, one with a phosphorus recovery efficiency of 54% as discussed in Blöcher et al. (2012) and another with 99% discussed in Gong et al. (2018). While the NUE of the fertilizer product recovered from WWTP1 is only 20%, that of the product from WWTP2 is 90%. Suppose both WWTPs have an inflow of 1000 kg P/month.
BA_{WWTP1} = 1000 × 0.54 × 0.20 = 108 kg
BA_{WWTP2} = 1000 × 0.99 × 0.90 = 891 kg

Through a combination of a high NUE and NRE, WWTP2 leads to a much higher BA compared to WWTP1.

4.3.3. Carbon cycling through soil organic matter addition

Rising greenhouse gas emissions have led to an increase in the average earth's surface temperature of 1.1 °C compared to the late nineteenth century (Viswanaathan et al., 2022). The negative effects of climate change in the form of extreme weather conditions such as more frequent heat waves, droughts, floods, and wildfires are evident. It is clear that even reaching net zero GHG emissions will only stabilize the warming and not reverse the damage nor eliminate the risks already caused by the risen temperatures (Rogelj et al., 2019). This proves the need for carbon sequestration.

Soil organic carbon (SOC) restoration can play a significant part in reversing climate change (Lehmann et al., 2020; Sommer and Bossio, 2014). About 26% of the SOC is estimated to have been lost from the top 30 cm of the soil globally due to land use changes (Sanderman et al., 2017) as a consequence of an increasing rate of organic matter volatilization due to higher temperatures (Lugato et al., 2021). Accordingly, the soil in the EU countries is declining in OM (Ferreira et al., 2022; Lugato et al., 2014). In the Netherlands, the trend of the soil organic matter (SOM) differs between regions (Hanegraaf et al., 2009).

The addition of SOM can sequester SOC subject to the local environmental conditions (Navarro-Pedreño et al., 2021). The most crucial factor for carbon sequestration is the stabilization of the OM (Navarro-Pedreño et al., 2021). For example, sewage sludge biochar can be stored in soil for a much longer time than untreated sludge (Zhao et al., 2023). The degree of stabilization of the wastewater-derived organic matter can vary significantly (Bożym and Siemiątkowski, 2018; Sánchez-Monedero et al., 2004), and must be accounted for. The SOM sequestration benefit is defined using the following equation:

$$SS = (1 - \frac{VS}{100}) \times OM_{soil} \tag{4.5}$$

where SS is the SOM sequestration in kg/y, VS is the volatile solids content of the recovered product (%) representing the labile carbon fraction, and OM_{soil} (kg/y) is the mass of organic matter applied to the soil.

Assuming 1000 kg of sludge produced by WWTP1 and WWTP2 is to be used for soil application. Whereas WWTP1 employs anaerobic digestion, WWTP2 employs aerobic digestion for sludge stabilization. The VS destruction achieved by both processes is assumed to be the same based on Metcalf & Eddy and AECOM (2014): at 50%. The OM percentages of the digested sludge products are 49.3% and 71.6% for WWTP1 and WWTP2 respectively as in Černe et al. (2019). Then, the SS values of the two WWTPs are as follows:

SSWWTP1 = (1-0.50) × 493 = 246.5 kg
SSWWTP2 = (1-0.50) × 716 = 358 kg

Thus, the WWTP2 achieves a larger SS due to a higher percentage (71.6%) of OM in the sludge product.

As shown in this section, the method can calculate three nature benefit indicators based on the mass flows of the recovered resources through the WWTP and the nature compartment where the resource is applied (freshwater streams, soil, or biomass). To ascertain the required mass flows, models or literature data can be used. A schematic diagram of the method is shown in Figure 4.2.

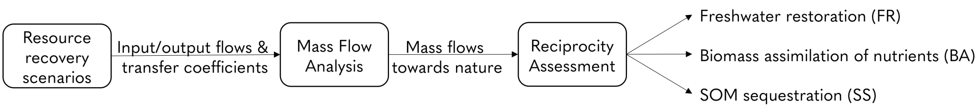

Figure 4.2: A schematic diagram explaining the connection between the resource recovery scenarios, the mass flow analysis of the resources and the reciprocity assessment indicators.

4.3.4. Case-study

To demonstrate the new method, an innovative WWTP planned for construction in Wilp, the Netherlands is used. The mass flows are based on a pilot study which is presented in Stowa (2023). To further understand the technical innovations of this treatment plant, the reader is directed to the same report.

Wilp treats the wastewater using predominantly physio-chemical processes in contrast to the biological processes most commonly employed in the Netherlands. Only the sludge is biologically treated using anaerobic digestion. This is a novel type of WWTP but, the method introduced is generally applicable. A schematic diagram of the WWTP is shown in Figure 4.3. As can be seen, the WWTP uses such processes as sieving, electro-coagulation (EC), dissolved air floatation (DAF), nanofiltration, and ion-exchange to recover the resources present in the wastewater while simultaneously meeting the national effluent quality standards. The WWTP concept has been successfully tested in a pilot. The treated effluent is assumed to be discharged indirectly into the IJssel River.

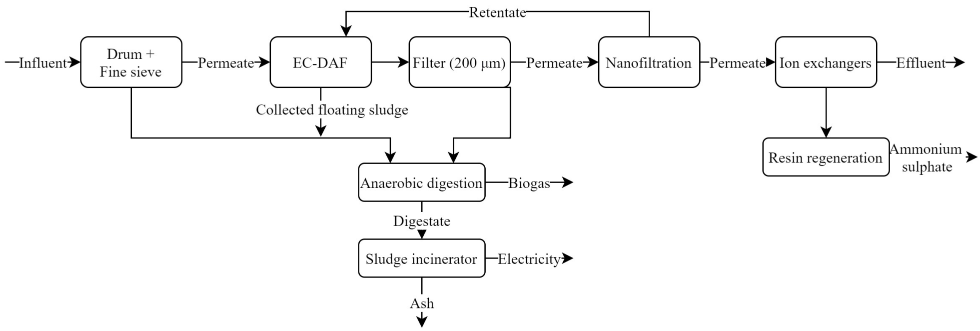

Figure 4.3: A schematic diagram of the Wilp WWTP based on the physio-chemical treatment of domestic wastewater (EC-DAF: electro-coagulation dissolved air flotation).

Currently, different recovery processes for P and organic matter are being explored in the Wilp WWTP. Thus, this study uses a base case of the WWTP with only the recovery of N as ammonium sulphate. The first scenario (Scn. 1) includes the recovery of N as ammonium sulphate, OM in the forms of cellulose fibres using a fine sieve and anaerobically digested sludge for soil application. The second scenario (Scn. 2) has an additional recovery of P in the form of struvite from the ash of the incinerated sludge. The third scenario (Scn. 3) includes the recovery of N as ammonium sulphate, OM as cellulose fibres and sludge digestate, and P recovery as vivianite using magnetic separation. These scenarios were chosen by the authors and the case-study owners. However, the method presented is generally applicable to other resource recovery scenarios.

Mass flows

The mass flows of organic matter, P, and N are calculated using the Substance flow analyser (STAN) (Cencic, 2008) software developed by TU Wien based on the information provided by the case-study owners.

First, the organic matter (COD) mass flows are described here. In the base case, about 30% of the influent COD is transferred from the water to the sludge phase using a drum and a fine sieve. The sieves mainly recover cellulose which constitutes about 30% of the influent COD (Reijken et al., 2018). Following this, about 22% of COD entering the EC-DAF process is separated from the water phase through coagulation. Further, a nanofiltration unit removes nearly 85% of the COD. The remaining COD in the water passes through the ion exchanger and gets discharged with the effluent. The sludge undergoes anaerobic digestion where about 65% of the influent COD gets converted into biogas according to Wan et al. (2016). The remaining COD is incinerated for energy production.

In Scn. 1, the cellulose is separated from the water through sieving and used in construction. The EC-DAF sludge undergoes anaerobic digestion which leads to the COD being transferred into a gas fraction containing CH_4 and CO_2 and a solid digestate which is assumed to be applied to agricultural land as soil amendment. The mass flows of the OM in this scenario (blue arrows) as well as the base case are shown in Figure 4.4.

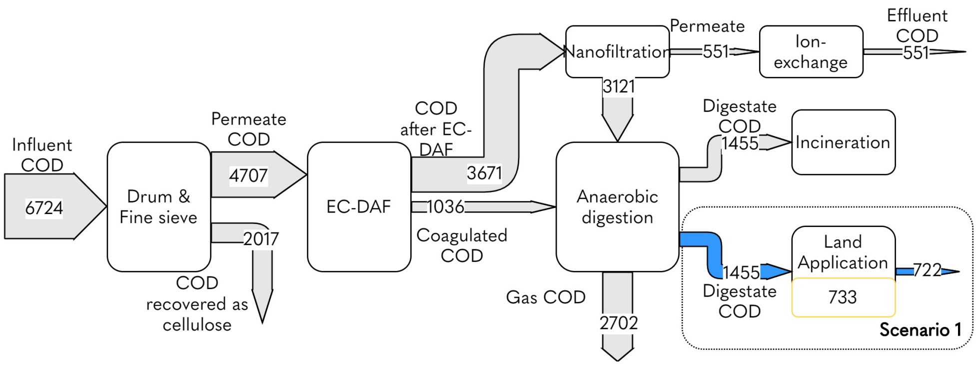

Figure 4.4: The mass flows ($\times 10^3$ COD kg/y) of organic matter through the Wilp WWTP in the base case and in Scn. 1. The blue coloured arrows represent Scn. 1.

Next, the mass flows of N are presented in Figure 4.5. Approximately 10% of the total nitrogen (TN) is removed by the nanofiltration process and 98 ± 2% is removed by the ion-exchanger. This removal efficiency of the ion-exchange process falls within the commonly cited range of 80-100% (Feng and Sun, 2015; Huang et al., 2020; Sica et al., 2014). The N is recovered in the form of ammonium sulphate upon the resin regeneration using sulphuric acid and is used as an agricultural fertilizer. The BA was calculated based on the average NUE of N fertilizers in Dutch agriculture, which is approximately 48% (CBS, 2022).

Next, the P mass flows are described and visualized in Figure 4.6. In the base case, most of the P is removed from the water through the EC-DAF process. A 98 ± 1% recovery efficiency is achieved for P using the EC-DAF process which falls within the 97-98% range reported in the literature (Bhoi et al., 2023; Inan and Alaydin, 2014; Yang et al., 2022). The recovered P is part of the sludge and passes through the anaerobic digestion process. About 10% of the P is assumed to be discharged with the digestion supernatant. 90% enters the sludge incinerator with the digestate. Nearly 100% of the P in the digestate ends up in the incinerator ash (Petzet et al., 2011) which is used in road construction.

In Scn. 2, along with the OM recovery, the P is recovered as struvite ($NH_4MgPO_4 \cdot 6H_2O$). Firstly, the influent P is incorporated into sludge using the EC-DAF process. 99% of the P present in the sludge gets transferred to the anaerobic digestate which goes to an incinerator. From the incinerator ash, acid leaching is used to recover struvite. A recovery efficiency between 80% to 95% can be found in the literature (Krüger et al., 2014; Petzet et al., 2011; Xu et al., 2012). In this study, a recovery efficiency of 90 ± 5% is assumed. The NUE of struvite is assumed to be 80% which is the average for P-fertilizers in Dutch agriculture (CBS, 2022).

In Scn. 3, instead of recovering P from the sludge ash, magnetic separation is used to recover vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$) from the digestate. The efficiency of P recovery as vivianite using magnets is about 60-64% of the total influent P (Wijdeveld et al., 2022). Here, a recovery efficiency of 64 ± 5% was assumed. Although vivianite has been reported to have a lower NUE compared to struvite (Ayeyemi et al., 2023), more research is needed to draw stronger conclusions. For now, an uptake efficiency equal to struvite (i.e., 80%) was assumed.

Nature reciprocity assessment

Freshwater restoration

For all three scenarios, the FR is equal because the effluent quality remains constant. To calculate the FR, the GWF is estimated for the different pollutants. While the GWF of the COD is 1.18×10^7 m^3/month, those of TN and TP are 3.85×10^5 m^3/y and 4.81×10^5 m^3/y respectively. Hence, the organic matter was found to have the largest GWF. This GWF was divided by the monthly streamflow of the River IJssel. River Ijssel is a distributary of the River Rhine (Hurkmans et al., 2022) and hence their discharges are correlated. The IJssel discharge was obtained using the Rhine discharge (based on Booij (2017)) and the empirical relationship of Hurkmans et al. (2022). The ratio between the monthly GWF of organic matter and the monthly river runoff was calculated. Based on this monthly WPL and TW discharge, the FR was calculated using Equation 4.3.

Biomass assimilation of nutrients

To calculate the nutrient assimilation, the removal efficiencies of the different treatment steps were obtained from the case-study owners and verified using the literature. The inflowing N and P masses were calculated by multiplying the influent wastewater volume with the nutrient concentrations. 6.07×10^5 kg N/y and 1.11×10^5 kg P/y were the inflowing mass flows of the nutrients. Based on the MFA, the recovery efficiency of N was found to be 88% under all scenarios. The recovery efficiency of P (struvite) was found to be 79% in Scn. 1. In Scn. 2, a 63% recovery efficiency was found for P (vivianite). Further, the NUE of N and P were assumed to be 48% and

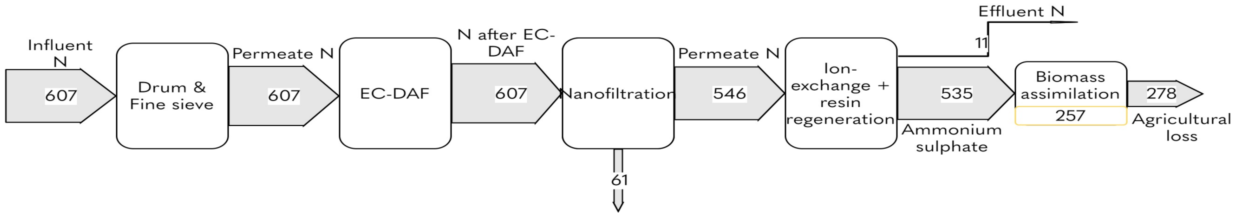

Figure 4.5: The annual mass flow (×10^3 kg TN/y) of total nitrogen (TN) through the Wilp WWTP (EC-DAF: electro-coagulation dissolved air flotation).

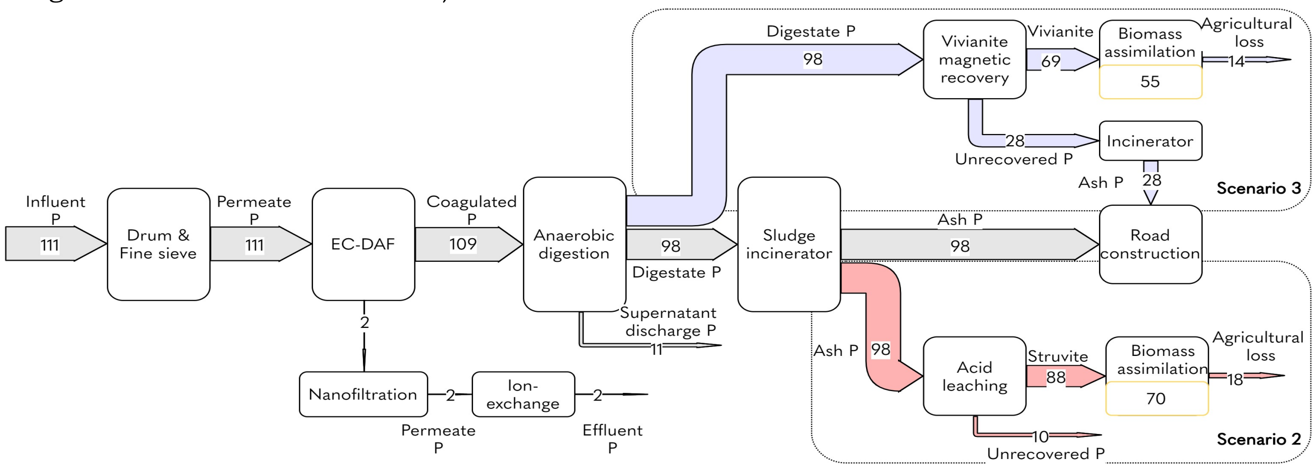

Figure 4.6: The annual flow of phosphorus (×10^3 kg TP/y) through the Wilp WWTP. The Scn. 3 flows are shown in light blue and the red coloured arrows show the flows of Scn. 2.

80% respectively based on CBS (2022). Thereafter, the BA of nutrients was estimated for N and P separately using Equation 4.4.

Soil organic matter sequestration

Here, it was assumed that the volatile organic components would be lost in a short time upon soil application. Therefore, only the non-volatile organics were assumed to be sequestered. The volatile component remaining in the sludge after anaerobic digestion was estimated using the Liptak equation that estimates the volatile solids reduction (Dagnew and Parker, 2021; Metcalf & Eddy and AECOM, 2014) as shown below:

$$VS_{reduction} = 13.7 \times Ln(SRT) + 18.9 \tag{4.6}$$

where SRT is the solid retention time of the anaerobic digester. An SRT of 10 days was assumed and using Equation 4.6 a 50.4% reduction in the volatile solids was found. The quantity of OM that can be applied to soil was estimated to be 1.45×10^6 kg/y using the MFA. The SS was calculated using Equation 4.5. For detailed calculations, the reader is directed to the Supplementary material (S4.1-4.7). Note that the calculations made here are specific to this case-study, this would be different for other technologies such as aerobic digestion, composting, and incineration.

Uncertainty analysis

The uncertainty in the reciprocity indicators could be caused by several factors. The exact NUE will depend on a lot of parameters including the farming practices. The VS in the soil application product can be determined experimentally but often may be estimated by equations which introduce certain uncertainty. Likewise, the recovery efficiencies used in the case-study were based on a pilot study and can vary for the full-scale plant. Therefore, an uncertainty analysis was conducted to provide a range for the reciprocity indicators based on the variation in the input parameters. In STAN, all uncertain inputs are normally distributed with a mean and a standard deviation that can be specified by the user (Laner et al., 2014). The mean entered usually originates from literature or an educated guess and not from a data sample, making the nature of the uncertainty epistemic and not random. Consequently, STAN converts the entered standard deviation into the standard error of the mean (SEM) using the following equation.

$$\sigma_X = \frac{\sigma}{\sqrt{N}} \tag{4.7}$$

where σ_X is the standard error of the mean, σ is the standard deviation specified by the user, and N is the number of data points. Since only the lower and the upper boundaries of the transfer coefficients are specified here based on literature and an educated estimation of the case-study owners, the number of data points (N) in this study is 2. STAN then makes use of the Gaussian error propagation (GEP) method for calculating the resulting uncertainties (Laner et al., 2014). For more details about the GEP method, the readers are directed to Lo (2005).

Sensitivity analysis

A sensitivity analysis was used to evaluate the effect of changing the parameters, such as the NRE and the river discharge. For the FR sensitivity, 20% lower and 20% higher COD loads in the WWTP effluent were used. Also, 20% higher and 20% lower river discharges were evaluated. To analyze the BA sensitivity, the N recovery efficiency of 88% estimated by the case-study owners was changed by 10% in both directions. A range of N and P recovery and uptake efficiencies were used, as shown in the Supplementary material S4.7. Lastly, for analysing the SS sensitivity, the VS content was modified to 30% and 70%, along with the original value of 49.6%.

4.4. Results

4.4.1. Reciprocity indicators

The nature reciprocity indicators along with their uncertainties are presented in Table 4.1. The FR remains the same for the four scenarios and is equal to 7.5×10^6 m^3/y because the effluent concentrations remain constant. The OM resulted in the highest GWF (1.2×10^7 m^3/month). However, compared to the runoff of the river IJssel (9.8×10^8 m^3/month on average), the GWF of the Wilp effluent was insignificant. Consequently, the WPL was very small. Therefore, a large portion (98%) of the discharged water (7.5×10^6 m^3/y) can be considered as freshwater restored into the river.

Through the recovery of ammonium sulphate and its application in Dutch agriculture, a biomass N assimilation of 2.6×10^5 kg/y is achieved. Since only one pathway of N recovery is used, the biomass N assimilation remains the same for all the scenarios. In the base case, no P is recovered and consequently, the biomass assimilation of P is 0. In Scn. 2, the biomass assimilation of P was found to be $7.0 \pm 0.3\times10^4$ kg/y. In comparison, a value of $5.6 \pm 0.3\times10^4$ kg/y was estimated for Scn. 3. Thus, the recovery pathway and the form of P recovered (struvite or vivianite) can substantially affect the biomass assimilation.

In the base case, organics were not recovered and thus the SS is equal to 0. In the other three scenarios, the SS was $7.3 \pm 0.8\times10^5$ kg/y. The SS for the base case was found to be 0 because the OM was partly used for biogas production and the rest was incinerated with the ashes being used in road construction. In the other three scenarios, part of the COD was recovered as cellulose fibres and used in construction. This part of the COD did not contribute to the SS. Another part of the COD was converted to biogas which also did not contribute. However, the digestate applied to the soil led to the sequestration of about 50% of the total OM. Considering nature benefits, Scn. 2 is the preferred scenario among the four. This is because it provides the same FR and BA of N as the other alternatives and

Table 4.1: Wilp WWTP base case compared to three different resource recovery scenarios. Scn.=Scenario.

Reciprocity indicators	**Base case**	**Scn. 1**	**Scn. 2**	**Scn. 3**
Freshwater restoration ($\times 10^6$ m^3/y)	7.5	7.5	7.5	7.5
Biomass assimilation of N ($\times 10^5$ kg/y)	2.6	2.6	2.6	2.6
Biomass assimilation of P ($\times 10^4$ kg/y)	0	0	7.0 ± 0.3	5.6 ± 0.3
Soil organic matter sequestration ($\times 10^5$ kg/y)	0	7.3 ± 0.8	7.3 ± 0.8	7.3 ± 0.8

leads to an SS equal to Scn. 1 and Scn. 3. However, Scn. 2 provides the highest BA of P.

Figure 4.7 visualizes the reciprocity of the base case and the three alternative scenarios using a spiderweb chart. The indicator value of each scenario has been normalized to that of the best performing one in that category (e.g., Scn. 2 for the biomass assimilation of nutrients) which is represented by 100.

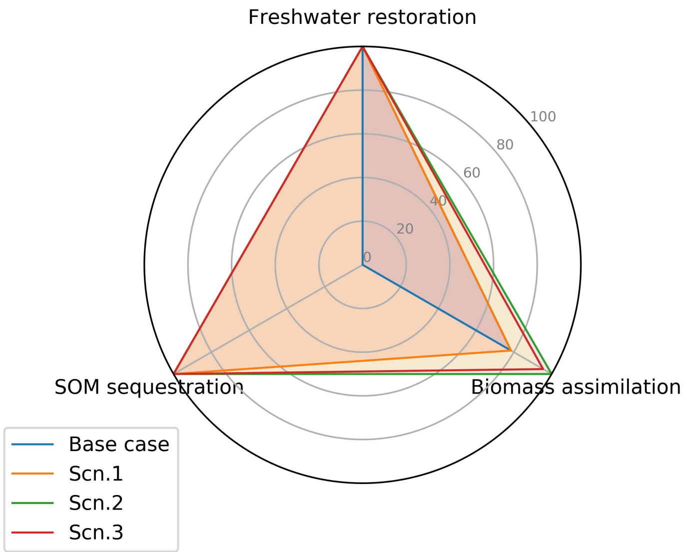

Figure 4.7: A comparison of the four scenarios on their nature reciprocity performance. The reciprocity values of the alternatives have been normalized relative to the highest value for that indicator (represented by 100).

The reciprocity indicators can be contrasted with the environmental damage type indicators commonly used. Here, the authors use the study of Tarpani et al. (2020) to contrast the two types of assessments. Tarpani et al. (2020) conducted an LCA to compare treatment methods to recover sewage sludge for different applications. They found the climate change potential of applying anaerobically digested sludge to agricultural soil to be -174 kg CO_2 eq./1000 kg DM. This value results from adding the CO_2 emissions of the electricity used in the anaerobic digestion and the CH_4 emissions after the digested sludge is applied to soil and subtracting the avoided burden of manufacturing industrial fertilizers. The avoided climate change impact of manufacturing fertilizers outweighs the impact of digesting the sludge and applying it to the soil, which led to a negative value for the climate change potential.

However, while an avoided burden can lead to negative values, this is distinct from the physical removal of pollutants from the environment (Tanzer and Ramírez, 2019). The negative value from avoided burden represents a potential reduction in carbon emissions. In contrast, the SS benefit indicator represents the carbon in the wastewater that is sequestered in the soil. While the LCA indicator measures the reduced environmental damage (a result of reduced greenhouse gas emissions), the benefit indicator measures a positive effect on the carbon cycle and the soil environment by physical SOM sequestration.

The reciprocity assessment can lead to different conclusions than an LCA. For example, there could be a treatment option with a higher energy/resource use that is able to contribute much more positively to nature e.g., implementing the nutrient recovery technologies are known to usually increase the global warming potential of WWTPs (Pausta et al., 2024; Pradel and Aissani, 2019) but they also enable the restoration of nutrient cycles which needs to be included in the sustainability discussion. In a study by Xu et al. (2014), anaerobic digestion followed by incineration was found to have a lower negative environmental impact compared to agricultural application of the digested sludge. However, basing a decision solely on lowering the negative environmental impacts may lead to ignoring the potentially positive effects of the agricultural application of the digested sludge in that case. Including the positive effects in the conversation may lead to the decision of using the digested sludge for agricultural application while trying to reduce the negative impacts of doing this (that is revealed by the LCA). In this way, the reciprocity assessment proposed here can be seen as a complementary tool to the LCA.

4.4.2. Sensitivity analysis

In Table 4.2, the percentage changes in the BA values are shown when the nitrogen recovery efficiencies and the nitrogen uptake efficiencies are changed from the case-study values of 88% and 48%. This is an example and the complete sensitivity analysis can be found in the Supplementary material S4.7. Modifying the COD load of the effluent by 20% and the River IJssel discharge by 20% did not have any significant effect on the FR of the WWTP. This was because of the large flow of the River Ijssel (1.2×10^{10} m^3/y) in comparison to the WWTP discharge (7.6×10^6 m^3/y).

For N (as shown in Table 4.2), a recovery efficiency value of 88% is al-

Table 4.2: The sensitivity of the biomass assimilation of N to changes in the nitrogen uptake and recovery efficiencies. The percentage changes are relative to the case-study values of 88% recovery efficiency and 48% uptake efficiency.

		NUE			
		30%	**48%**	**60%**	**80%**
	78%	-45%	-11%	11%	48%
NRE	**88%**	-38%	0%	25%	67%
	98%	-30%	11%	39%	86%

ready high. Improving it to 98% can lead to an improvement in the BA by about 11%. However, a much higher improvement in the BA can be achieved by increasing the NUE, which lies around 48%. Whereas increasing the NUE to 60% can lead to an increase in the BA by 25%, increasing it to 80% (equal to that of P), can lead to a BA improvement of 67%. An 86% increase in BA is possible when both a high N recovery efficiency of 98% and a high N uptake efficiency of 80% are achieved.

For P, which already has a high NUE in Dutch agriculture (80%), improving the NUE to 90% has a limited effect on the BA, improving it by 13%. On the other hand, improving the P recovery efficiency from the WWTP from 79% to 90% can improve the BA by 43%. High recovery and uptake efficiency values of 90% can improve the BA by 61%. Analysing the sensitivity of the soil carbon sequestration values, decreasing the VS content of the soil amendment products by 20% can improve the SS by 40%.

4.5. Discussion

4.5.1. Reciprocity assessment

The innovative method helps quantify potential positive effects on the natural environment from the resources recovered from WWTPs using a life cycle approach. The indicators are calculated using parameters related to a WWTP (e.g., recovery efficiency) and also those related to the application of the resources (e.g., NUE for nutrients). This will encourage decision-makers to think about the resource recovery solutions down to the application process and thereby prevent burden-shifting. Moreover, the assessment relies on data such as recovery efficiencies and OM content that are easily available to decision-makers. Thus, a major advantage of this method is the ease of calculation. Certainly, more complex models can be developed to calculate the indicators but, the method captures sufficient details to differentiate between the different resource recovery options (e.g., vivianite or struvite recovery). The different resources scenarios that the Wilp WWTP adopts can notably vary the kind and extent of the nature benefits.

The method was demonstrated on a WWTP that relies mostly on physio-chemical treatment. However, WWTPs can have a variety of configurations and the method proposed here can easily be applied to these as well. This is because the method is generic, i.e., independent of the treatment process involved and can be used as long as the mass flows of the relevant resources can be calculated.

Wilp can restore 7.5×10^6 m^3/y (92% of the influent) of freshwater into the IJssel River. The discharge water had a very low WPL (monthly average of 1.2×10^{-2}). Consequently, a negligible portion of the annual streamflow is required to dilute this effluent. Therefore, the FR achieved by the WWTP is almost equal to the discharged effluent. Two important observations should be made. First, the low WPL shows that focusing on the recovery of OM and nutrients from wastewater leads to effluent quality with a very low WPL. Second, the high streamflow rate (9.8×10^8 m^3/month on average) of the IJssel River is an important factor leading to a high FR value. The FR values were sensitive to neither the COD load in the WWTP effluent nor the Ijssel flow rate (± 20%). High removal rates of the organic matter and the nutrients help to restore natural stream flows and maintain their quality. Additionally, owing to the high effluent quality, other high-value applications of the effluent may be considered such as managed aquifer recharge.

4

Using the recovered N from Wilp, 2.6×10^5 kg TN/y can be assimilated into plant biomass. Furthermore, $7.0 \pm 0.3\times10^4$ kg TP/y and $5.6 \pm 0.3\times10^4$ kg TP/y can be assimilated in scenarios 2 and 3. The nutrients excreted by humans would be dissipated in the natural environment (soil and water bodies) unless collected from the domestic sewage and recycled. By actively sequestering the nutrients into plant biomass, a WWTP can provide a crucial nature benefit.

The BA achieved by a WWTP depends on the efficiency of the nutrient recovery technology and the nutrient uptake of the fertilizer. The P recovery from the ash after incineration offers a higher recovery efficiency (80%) compared to the vivianite recovery using magnetic separation (64%). Furthermore, struvite fertilizers have a higher P uptake efficiency and thus contribute to better cycling of nutrients compared to conventional P fertilizers (Li et al., 2019; Uysal et al., 2014). Therefore, to remove the excess reactive nutrient species from the natural environment, two factors are essential. The NRE as well as the NUE of the recovered nutrients must be high. Furthermore, to increase the BA of N, improving the N uptake efficiency in agriculture should be the focus. In contrast, P already has a relatively high uptake efficiency and thus the focus should be more on achieving high recovery efficiencies. This study showed that struvite recovery from sludge ash is the most promising pathway from the perspective of biomass assimilation, as also noted by Egle et al. (2016).

SOM restoration can improve desirable soil properties and also help to sequester carbon to mitigate climate change. However, in the Netherlands, there is no clear trend in the SOM and careful consideration is needed to decide where to use the sludge-derived products. Assuming the location-suitability, an addition of $7.3 \pm 0.8 \times 10^5$ kg/y SOM can be achieved by Wilp after the anaerobic digestion of its sludge. From a climate change perspective, this number is likely to be insignificant to offset the CO_2 emissions. However, even a small addition of SOM can have a significant positive effect on the local environment and agricultural productivity in soils with declining organic content (Hanegraaf et al., 2009). Furthermore, a sludge stabilization method other than anaerobic digestion may retain more biodegradable organic content that can be applied to the soil. Decreasing the VS content of the sludge product can increase the carbon sequestered in the soil. Thus, where higher organic matter sequestration is required, a sludge stabilization process can be selected that retains more organic matter, such as lime stabilization (Yoshida et al., 2018).

It is important to clarify that providing nature benefits alone does not qualify a WWTP as sustainable. The chemical use, energy consumption, and emissions of the WWTPs are crucial factors to be considered and an LCA can help assess these. Wilp is predominantly a physio-chemical WWTP which has both disadvantages and advantages compared to a conventional activated sludge WWTP. An LCA conducted by Stowa (2023) reports that Wilp uses more electricity (2.5 times more) and higher dosages of chemicals than conventional activated sludge WWTPs. However, the Wilp WWTP also has zero direct emissions of CO_2 and N_2O, which implies that higher proportions of OM and N present in the wastewater can be used for benefiting nature through the mechanisms discussed in this chapter.

4.5.2. Limitations

Resources recovered from wastewater contain heavy metals, pathogens, and organic micropollutants that also damage the natural environment. This study ignores their presence because the focus was on assessing the nature benefits. For assessing the negative effects of such substances, damage units developed by Egle et al. (2016) can be used.

The nutrient uptake efficiency can vary widely based on the type of fertilizers, soil characteristics, climate, and agricultural practices. In this study, the average efficiency values were obtained for the Netherlands and a sensitivity analysis was used to cover a certain range of the uptake efficiencies. However, more studies are needed to quantify the improved nutrient use efficiencies of slow-release fertilizers obtained from wastewater.

In this study, one of the resource recovery pathways included the use of a sludge-derived product as a soil amendment. In the Netherlands, sewage sludge products are not applied to agricultural soils (Racek et al., 2020). However, this pathway was included because soil application of sludge (and its derived products) is practised in many parts of the world including other EU countries (Hudcová et al., 2019).

4.6. Conclusions

In this chapter, a novel method was developed to assess the potential nature benefits of the resources recovered from wastewater. The following conclusions can be drawn:

- The proposed method works well to quantify key benefits to the natural environment from wastewater-based resources from a life cycle perspective. In the planning and assessment of the resource recovery solutions, the focus should not be limited to reducing the negative environmental impacts. Instead, the nature benefits that can be obtained through the recovered resources should be included in the overall sustainability assessment.

- Focusing on maximizing the recovery of the organic matter and the nutrients in domestic wastewater can significantly improve the effluent quality. The discharge of treated effluent into a stream with a high dilution capacity due to high flow rates can help restore the quality and quantity of freshwater in nature.

- WWTPs also help to transform the waste nutrients into fertilizers with high uptake efficiencies, thus contributing towards more effective biomass assimilation of nutrients. This can help to reduce the reactive nutrient emissions below the planetary assimilation limits.

- WWTPs can also help to restore soil organic matter, which can mitigate climate change and improve soil quality. The stabilization of the organic matter achieved by the WWTP will decide the extent of sequestration. Including the reciprocity assessment in the decision-making process can help uncover the advantages of certain resource recovery pathways that are not yet in practice.

In conclusion, the method proposed in this study is a start towards recognizing and quantifying the potentially positive role of humans in the natural environment through resource recovery solutions.

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5

Water-food-energy framework to assess decentralized source separation

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5.1. Introduction

The water treatment sector is closely linked to the food and energy production sectors due to the exchange of the recovered resources. The assessment of the water, food, and energy resources should be done in an integrated manner and hence, the water-food-energy (WFE) nexus concept is discussed (El-Gafy, 2017). However, assessing the management practices of these three resources is a complex undertaking (Albrecht et al., 2018; Dargin et al., 2019) and reproducible assessment methods are lacking (Albrecht et al., 2018; Cairns and Krzywoszynska, 2016; Nhamo et al., 2020; Shannak et al., 2018).

Further, there are two main approaches to wastewater treatment and resource recovery: conventional centralized and decentralized source separation (DSS). The two approaches have been compared in numerous sustainability studies with mixed results (Firmansyah et al., 2021; McConville et al., 2017). Currently, the trend is to move from centralized treatment towards DSS or a hybrid of the two (Poustie et al., 2015). As this transition takes place, it is important to compare the two approaches using a WFE framework. This would allow one to evaluate the pros and cons of the two in terms of the water treatment sector as well as the closely-linked energy and food production sectors. Such a comparison is lacking to the best of the author's knowledge. This chapter will present a novel WFE framework to compare a conventional centralized and a DSS treatment plant in an integrated and holistic manner.

The chapter is organized as follows. The background of the WFE nexus is presented in Section 5.2, along with an introduction to the concepts of decentralization, source separation, efficiency, circularity, energy self-sufficiency, and nature reciprocity. In Section 5.3, the novel WFE framework is presented along with the assessment methods and the case-study descriptions. Section 5.4 contains the results of the comparison between the two case-studies. A discussion about the WFE framework's application to compare the two cases follows in Section 5.5. In Section 5.6, conclusions about the novel WFE framework, the comparison between DSS and conventional treatment approaches, and recommendations are presented. The calculations are presented in Supplementary material (S5.1-5.10)

5.2. Background

5.2.1. Decentralized source separation versus conventional centralized treatment

Although decentralization and source separation are two independent concepts, they are often discussed together and some authors believe that their combination is key to capturing the benefits of both (Opher and Friedler, 2016; Guest et al., 2009; Roefs et al., 2017). In this study, the two concepts are considered together in one system and thus, a conventional centralized

WWTP is compared to one with decentralized source separation.

There is a tendency to shift from centralized treatment towards DSS or a hybrid of the two (Mannina and Viviani, 2009). Conventionally, domestic wastewater is collected from homes and transported through sewers to a central WWTP before being treated and discharged (Bernal et al., 2021). However, the goal of WWTPs is no longer limited to treating the wastewater to discharge standards. Resource recovery is increasingly becoming an additional focus (Renfrew et al., 2022). For resource recovery, the conventional centralized approach may be adopted but a DSS treatment may offer distinct advantages.

Domestic wastewater is a mix of black water (BW), originating from the toilets, and grey water (GW), originating from washing activities (Tervahauta et al., 2014). BW carries approximately 90% of the N, 77% of the P, and 55% of the organic matter measured by COD (Roefs et al., 2017). DSS could refer to the separate collection of urine, or that of BW and GW using two different pipes or even the separate collection of urine, BW, and GW, the latter two options being more promising because of more resource recovery opportunities (Besson et al., 2021). Due to the differences in their compositions, while the GW is more suited for treated wastewater reuse because of low pathogen concentrations (Paulo et al., 2013), the BW can be targeted for energy and nutrient recovery (Pasciucco et al., 2022).

DSS can offer some advantages from the resource recovery perspective. Due to the lower dilution of organic matter and nutrients, resource recovery is more efficient from source-separated streams (Pasciucco et al., 2022). For example, Kjerstadius et al. (2017) found a higher nutrient recovery efficiency with a lower carbon footprint resulting from the separate GW and BW collection when compared to a mixed stream collection. Source separation also makes it possible to have smaller reactors to treat the different streams separately, making the treatment plants more compact and less complex and thereby saving on capital costs and land (Capodaglio, 2017; Opher and Friedler, 2016).

On the other hand, decentralized treatment often lacks adequate financial and technical support and suffers from diseconomies of scale, especially for energy use efficiency, and technological and political lock-in (McConville et al., 2017). The lock-in can be attributed to the narrative that a DSS approach needs to be profitable from the start and thereby can preclude the government to fund such a transition (Ampe et al., 2020). Additionally, source separation is often considered immature and risky by wastewater experts (Guest et al., 2009) as well as expensive to monitor (Diaz-Elsayed et al., 2019). Furthermore, high upfront capital costs for retrofitting toilets and installing extra pipelines prove to be barriers to the more widespread adoption of DSS (Diaz-Elsayed et al., 2019). Additionally, due to a lower dilution, the higher concentration of contaminants in the source-separated streams are a risk for sudden point source pollution and a threat to public health (Schoen and Garland, 2017).

Therefore, a DSS treatment has both advantages and disadvantages which are dependent on factors such as energy consumption, freshwater sources, and nutrient emissions (Opher and Friedler, 2016). These pros and cons need further study subject to different contexts and system boundaries (Lam et al., 2015; McConville et al., 2017). Further, the advantages of DSS may be more prominent when considering the other sectors where the resources recovered from a WWTP are used. Therefore, a WFE framework will serve the purpose of clearly demonstrating and quantifying the advantages of DSS. Yet, to the extent of the author's knowledge, no WFE framework has been used to compare a conventional centralized and a DSS treatment approach. This knowledge gap will be covered in this chapter.

5.2.2. The water-food-energy nexus

The water-food-energy (WFE) nexus refers to the complex links and dependencies between the three major resources and the need for cross-sectoral coordination for their utilisation (Smajgl et al., 2016). The need for a cross-sectoral perspective was felt as numerous interactions exist between the three sectors. Water is required to produce food (e.g., irrigation) but also for energy production (e.g., for cooling of power plants). Energy is used throughout food production and transportation as well as in water treatment (El-Gafy, 2017). Thus, water, food, and energy cannot be managed well in isolation and require an integrated approach (El-Gafy, 2017; Elsayed et al., 2020).

Most of the discussion around the WFE nexus remains theoretical (Albrecht et al., 2018). Although there is a limited number of frameworks addressing the three sectors (Shannak et al., 2018), an integrated approach to facilitate cross-sectoral coordination is needed (Nhamo et al., 2020). Fetanat et al. (2021) developed a decision-making WFE framework for energy recovery from WWTPs, but other resources, such as TW and nutrients, were not covered. Since the TW and nutrients are frequently exchanged between the three sectors, they should be included. Yi et al. (2020) and Simpson et al. (2022) developed composite indicators to measure and monitor the individual performances and the linkages between the three sectors at a national level and the provincial scales. However, this may not serve the decision-makers to design and monitor the resource recovery solutions at a WWTP scale. Furthermore, these frameworks lack the consideration of circularity indicators. The WFE nexus index method developed by El-Gafy (2017) included indicators to calculate the total water and energy consumption for food production but it does not account for the circularity and is focused on the food production sector. Thus, the few existing frameworks lack in some aspects.

Moreover, the economic dimension is insufficiently covered by most frameworks (Shannak et al., 2018). Also, these lack factors such as the local climate which will dictate the water requirement (e.g., for irrigation) (Shannak et al., 2018). The frameworks have also been criticized for a lack of practical applicability (Cairns and Krzywoszynska, 2016), and of analytical tools and reproducible methods to evaluate real-world cases (Albrecht et al., 2018; Nhamo et al., 2020).

Therefore, a new WFE framework with reproducible methods is presented in this chapter to compare a conventional centralized and a DSS-based approach to wastewater treatment and resource recovery. The novelty of this framework is that it offers an integrated and more holistic way that accounts for the links between the water, food, and energy sectors, allowing for a better comparison between the two approaches. The assessment is centred around a WWTP and is meant for the decision-makers to assess the resource recovery solutions at the scale of a WWTP. The methods of this framework will allow the decision-makers to evaluate how the implementation of a particular resource recovery solution can contribute positively to the food production sector subject to the specific water requirements of a particular region. This may also serve to improve the intersectoral communication which is often found wanting in the WFE nexuses (Greer et al., 2020). Further, a circularity assessment method will also be used as part of this framework as the current WFE frameworks usually lack circularity indicators.

5.3. Methods

5.3.1. WFE assessment framework

The WFE assessment framework developed here is presented in Figure 5.1. This framework is designed to keep the water treatment sector at the center but includes the food production and energy production sectors too. The assessment is done using indicators belonging to the three sectors and functional units of the water treatment and food production sectors. The functional unit for the water treatment sector is the treatment of a certain volume of wastewater to the relevant effluent standards using a WWTP. For the food production sector, irrigation and fertilization of the agricultural land is the functional unit. And, from the energy production perspective, self-sufficiency of the WWTP is the goal.

Efficiency assessment

Efficiency can be broadly defined as the ratio between useful outputs (benefits) and the inventoried flows (of resources, energy, money, etc.) or environmental impacts. Huysman et al. (2015) defined two levels of efficiency: Level 1 refers to the ratio between benefits and inventoried flows and level 2 is the ratio between the intended effects or benefits and the environmental impact (eco-efficiency). In this paper, level 1 efficiency indicators will be used. This is because this framework is meant for decision-makers who may not have the necessary skills and resources to conduct an LCA. If necessary and in the case that LCA results are available, they can be easily included as denominators in the efficiency formulae. The efficiency indicators will be expressed as the ratio between the functional unit of a pro-

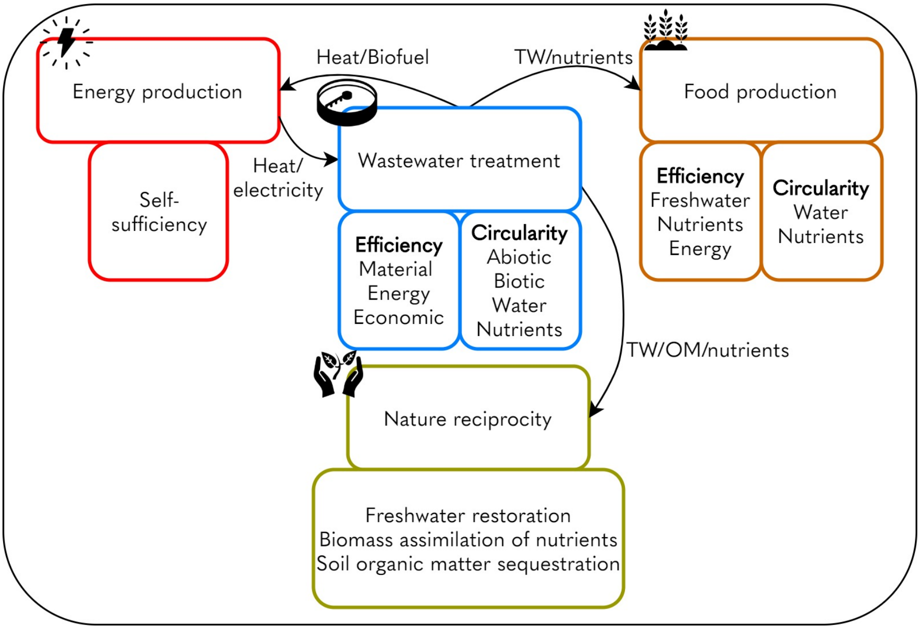

Figure 5.1: The water-food-energy nexus assessment framework of assessment methods and indicators.

cess (the useful output) and the inventoried flows (material, energy, and economic investment). To illustrate, the energy efficiency of the water treatment sector will be defined as the ratio between the volume of wastewater treated (m^3/y) to the energy consumed (kWh/y). A higher efficiency value is naturally more desirable. For the material efficiency indicators, the ratio between the functional unit of a process and the linear flowing material flows (kg-linear/year) of the process is used. The linear flow is obtained as part of the material circularity indicator calculation, which is explained in depth in Bhambhani et al. (2023).

Efficiency is calculated separately for the water treatment and food production sectors, using their respective functional units as numerators.

Water treatment: The functional unit is the annual volume of wastewater treated (m^3/year) per person equivalent (p.e.) to meet local effluent standards.

Food production: The functional unit is defined as the irrigation and fertilization of arable land required to support the same p.e. Here, the arable land area per person, based on The world bank (2024), and the most cultivated EU crop, common wheat (Eurostat, 2024), are used to estimate water and nutrient needs. These requirements can also be calculated for other crops if necessary.

The authors focus on the inventoried flows most relevant to the interaction between the water treatment and food production sectors. For the water treatment sector, these include material, energy, and economic costs (Capex and Opex). For the food production sector, freshwater, nutrients, and energy are considered. Although the centralized WWTP is already operational, the Capex pertains to upgrades aimed at improving resource recovery, detailed in the case-study section. Based on the above-mentioned functional units and inventoried flows, the following indicators are used:

Material efficiency ($E_{WT\text{-}Mat}$ in m^3/kg-linear): The ratio of the annual wastewater volume treated (V_{WW}, m^3/y) to the linear mass flow of resources through the WWTP. The linear mass flow is calculated as the product of the mass of resources used (M_{in}, kg/y) and a linear flow indicator (1 - MCI_{mixed}).
Energy efficiency ($E_{WT\text{-}En}$ in m^3/kWh): The volume of wastewater treated annually per unit of energy input to the WWTP (E_{in}, kWh/y).
Economic efficiency ($E_{WT\text{-}Econ}$ in m^3/€): The annual wastewater volume treated divided by the total annualized costs (€/y), including capital and operational expenditures of the WWTP.

5

For the food production process, efficiencies are calculated as follows:

Freshwater efficiency ($E_{FP\text{-}FW}$ in m^2/m^3-linear): The annual irrigated land area (A_{agri}, m^2/y) divided by the linear flow of irrigation water, expressed as the product of the irrigation water volume (V_{irr}, m^3/y) and the linear flow indicator (1 - MCI_{FW}).
Nutrient efficiency ($E_{FP\text{-}Nut}$, m^2/kg-linear): The annual irrigated land area divided by the linear flow of nutrients, calculated as the product of the nutrient mass used (M_{Nut}, kg/y) and the linear flow indicator (1 - MCI_{Nut}).
Energy efficiency ($E_{FP\text{-}En}$, m^2/kWh): The annual irrigated land area divided by the energy consumed for irrigation (E_{irr}, kWh/y).

The efficiency equations are shown in Table 5.1 along with the circularity, nature reciprocity, and energy self-sufficiency ones.

Circularity assessment

The circular economy is an alternative to the current linear economy that aims to recirculate resources within the economic production system to maximize the recovery of value (Corona et al., 2019). Circularity is a measure of the extent to which the circular economy has been implemented, in other words, the extent to which virgin resource extraction and unrecovered waste generation are avoided (Ellen MacArthur Foundation, 2019). Circularity may be assessed at the level of the economy but here the focus is on the water treatment and food production sectors. Several circularity assessment methods have been developed to measure the decoupling of economic progress from resource depletion (Rigamonti and Mancini, 2021). In this chapter, the modified material circularity indicator (MCI) from Bhambhani et al. (2023) will be used. This method is based on the original MCI developed by the Ellen MacArthur Foundation (2019) and it helps to assess the percentage of the total material resource throughput of a process that is circular.

Here, the circularity assessment measures the percentage of the total resource flows through the water treatment and the food production processes that are circular and the focus is on resources that are relevant to the WWTP and/or exchanged between the two sectors. These include mixed resources which refer to any resource (precipitation chemicals, acids, bases, biochar, etc.) being used in the water treatment process. Further, the circularity values are calculated for biotic resources, nutrients, and water for the water treatment. For food production, the circularities of freshwater and nutrients are calculated. The MCIs are calculated as follows:

Virgin input (V in kg/y): The fraction of resource input that is restorative in nature (RSIF) is subtracted from 1 and multiplied by the total resource input (M in kg/y).
Unrecovered waste (W in kg/y): The fraction of restorative and regenerative output flows (RSOF and RGOF) is subtracted from 1 and multiplied by the total resource input (M in kg/y).
Linear flow indicator (LFI): The sum of W (kg/y) and V (kg/y) is divided by two times the total resource input (M in kg/y).
Material circularity indicator (MCI): The LFI is subtracted from 1 to give the MCI.

Nature reciprocity assessment

Nature reciprocity can be defined as the quantification of the positive effects on the natural environment through a re-balancing of resource stocks (Bhambhani et al., 2024). The potential as well as duty of human society to actively benefit the natural environment has been ignored in conventional sustainability discourses for a long time (Bhambhani et al., 2024). In this framework, this potential will be taken into account and assessed using the nature reciprocity indicators. Nature reciprocity here refers to the re-balancing of the stocks of freshwater, N, P, and organic matter that can be achieved using the resources recovered from WWTPs.

The three nature benefits assessed are linked to the water, nutrient, and carbon cycles, measured using freshwater restoration (FR), biomass assimilation of nutrients (BA), and soil organic matter sequestration (SS).

Freshwater restoration: This measures the quantity of freshwater a WWTP returns to the environment via treated wastewater discharge, adjusted for effluent quality. It is calculated as the WWTP's monthly discharge volume (Q_{dis} in m^3/month) minus the stream flow fraction needed to dilute the effluent, determined by its water pollution level (WPL).

Biomass assimilation of nutrients (BA): This indicator quantifies the nitrogen (N) and phosphorus (P) cycled back into the environment via biomass. It factors in the WWTP's nutrient recovery efficiency (NRE), the nutrient uptake efficiency (NUE) of recovered products, and the nutrient inflow to the WWTP (M_{inf} in kg/y).
Soil organic matter sequestration (SS): This measures the organic matter sequestered in soil, based on the total organic matter applied (OM_{soil} in kg/y) and its stability, evaluated by the volatile solids content (VS%).

These equations are presented in 5.1, and for a detailed explanation of these indicators, the readers are referred to Bhambhani et al. (2024).

Energy self-sufficiency assessment
Energy self-sufficiency is here defined as the quantity of energy recovered (kWh) from a WWTP (heat+electricity) expressed as a percentage of the energy used (kWh) by the WWTP. This is the definition used by several authors when discussing the concept of energy self-sufficiency including Maktabifard et al. (2018); Wang et al. (2016); Yan et al. (2017), among others. This is one way to measure the progress of a WWTP towards sustainability concerning energy use and it will be used as part of the novel framework.

Water treatment and agriculture account for up to 5 % of the total electricity demand of some countries (Longo et al., 2016). WWTPs require energy to run the treatment processes and the requirement varies according to factors such as the processes, the location, pollutant loads, environmental standards, and the infrastructure age (Maktabifard et al., 2018). The energy use is expected to grow in the coming years (Yan et al., 2017). However, energy can also be recovered from WWTPs in various ways. Whereas, anaerobic digestion of excess sludge can yield biogas that can be used to produce electricity, heat exchangers and heat pumps may be used to recover thermal energy. Further, microbial fuel cells can convert the organic energy present in the wastewater directly to electricity (Wang et al., 2016). This has led to the discussion of energy-self-sufficient/energy-neutral WWTPs. The indicator is calculated as follows:

Energy self-sufficiency (%): The ratio between the energy recovered (kWh/y) and the energy used by a WWTP (kWh/y) multiplied by 100.

Therefore, 100% energy self-sufficiency implies that the WWTP can theoretically supply all of its energy requirements. If the WWTP can produce more energy than it needs, then the indicator value will be more than 100% and if no energy is produced by the WWTP, the value will be 0%. In the latter case, all of the energy requirement of the WWTP has to be externally sourced. It is important to note that even if the energy produced at the WWTP is used for a purpose unrelated to the WWTP (e.g., transportation fuel), it is still counted. The indicators for the efficiency, circularity, nature reciprocity and energy self-sufficiency assessment are listed in Table 5.1.

Table 5.1: Indicators used in the novel WFE Framework. V = virgin mass; W = unrecovered mass; RSIF = restorative input fraction; RSOF = restorative output fraction; RGOF = regenerative output fraction; M = mass of resources; MCI = material circularity indicator; LFI = linear flow indicator; Q_{dis_i} = treated wastewater discharge in month i; WPL_i = water pollution level in month i; NRE = nutrient recovery efficiency; NUE = nutrient uptake efficiency; VS = volatile solid component; $\mathrm{OM}_{\mathrm{soil}}$ = organic matter added to the soil; E_{rec} = Recovered energy; E_{in} = Input energy.

Water treatment (WT)	
Circularity (mixed, biotic, nutrients, & water)	V=M(1-RSIF) W=M(1-RSOF-RGOF) LFI=$\frac{V+W}{2M}$ MCI =1-LFI
Material efficiency	$\mathrm{E}_{\text{WT-Mat}}(\mathrm{m}^3/\text{kg-linear})=\frac{V_{\mathrm{WW}}(m^3/y)}{M_{\mathrm{in}}(kg/y)\times(1-MCI_{\mathrm{mixed}}}$
Energy efficiency	$\mathrm{E}_{\text{WT-En}}(\mathrm{m}^3/\mathrm{kWh})=\frac{V_{\mathrm{WW}}(m^3/y)}{E_{\mathrm{in}}(kWh/y)}$
Economic efficiency	$\mathrm{E}_{\text{WT-Econ}}(\mathrm{m}^3/€)=\frac{V_{\mathrm{WW}}(m^3/y)}{Capex+Opex(€/y)}$
Food production (FP)	
Circularity (water & nutrients)	V=M(1-RSIF) W=M(1-RSOF-RGOF) LFI=$\frac{V+W}{2M}$ MCI =1-LFI
Freshwater efficiency	$\mathrm{E}_{\text{FP-FW}}(\mathrm{m}^2/\mathrm{m}^3\text{-linear})=\frac{A_{\mathrm{Agri}}(m^2/y)}{V_{\mathrm{irr}}(m^3/y)\times(1-MCI_{\mathrm{FW}})}$
Nutrient efficiency	$\mathrm{E}_{\text{FP-Nut}}(\mathrm{m}^2/\text{kg-linear})=\frac{A_{\mathrm{Agri}}(m^2/y)}{M_{\mathrm{Nut}}(kg/y)\times(1-MCI_{\mathrm{Nut}})}$
Energy efficiency	$\mathrm{E}_{\text{FP-En}}(\mathrm{m}^2/\mathrm{kWh})=\frac{A_{\mathrm{Agri}}(m^2/y)}{E_{\mathrm{irr}}(kWh/y)}$
Nature reciprocity	
Freshwater restoration	$\mathrm{FR} = \sum_{i=1}^{12}\left(Q_{\mathrm{dis}_i} \times (1-\mathrm{WPL}_i)\right)$
Biomass assimilation of nutrients	$\mathrm{BA}=\mathrm{M}_{\mathrm{inf}} \times NRE \times NUE$
Soil organic matter sequestration	$\mathrm{SS}=(1-\frac{VS}{100}) \times OM_{\mathrm{soil}}$
Energy production (EP)	
Energy self-sufficiency	$ESS = \frac{E_{\mathrm{rec}}(kWh/y)}{E_{\mathrm{in}}(kWh/y)}$

5.3.2. Case-studies

A brief description of the two case-studies is presented below. It must be noted that the semi-hypothetical case-studies used here are based on two real-life cases but contain a few assumptions and simplifications.

Corleone: a centralized conventional WWTP

A flowchart of this case-study is shown in Figure 5.2. This is an activated sludge WWTP treating 3700 m^3/d of domestic wastewater and is designed for 12000 p.e. (Mannina et al., 2022). The WWTP is supplied with an intermittent aeration (IA) system to reduce energy use. Furthermore, an oxic settling anaerobic (OSA) reactor is present to reduce the excess sludge quantity. Some of the water will pass through an ultrafiltration (UF) unit to produce irrigation water (Mannina et al., 2022). A single UF module is currently operational and has a capacity of 25 m^3/h thus limiting the quantity of irrigation water that can be supplied. Since this WWTP is a conventional one, the authors here have assumed that its distance from the agricultural fields is 2 km on average. This is an arbitrary choice since no data was available and a sensitivity analysis is conducted. It must be noted that the excess sludge is land-filled. In the future, it is expected to be composted and the compost to be used in agriculture. The composting process is included in this case-study.

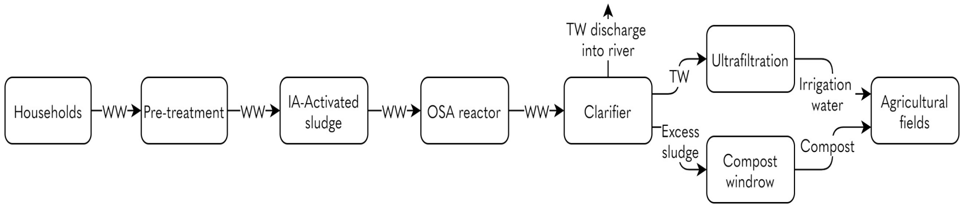

Figure 5.2: Flowchart depicting the Corleone (conventional) case-study. WW-Wastewater, TW-Treated wastewater, IA-Intermittent aeration, OSA-Oxic settling anaerobic.

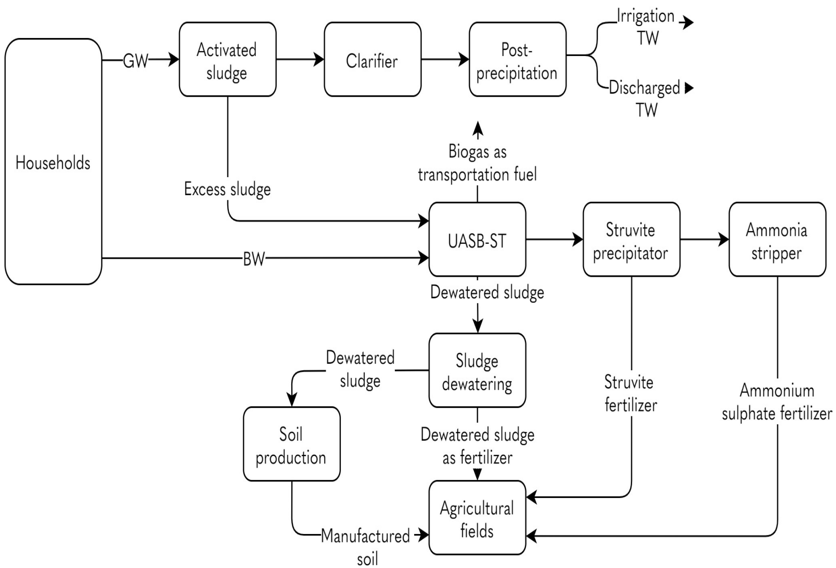

Figure 5.3: Flowchart depicting the Helsingborg (source separation) case-study. GW-Grey water, BW-Black water, TW-Treated wastewater, UASB-ST-Upward anaerobic sludge blanket-septic tank.

Helsingborg: a DSS WWTP

The Helsingborg case-study, presented in Figure 5.3, consists of a DSS WWTP with a capacity of 12000 p.e. In reality, each house has 3 pipes; one for BW, another for GW, and the last one for food waste collection. For the case-study in this chapter, the food waste collection and treatment has been excluded from the system boundary and only the wastewater flows are considered for a fair comparison with Corleone.

The BW is collected from the vacuum toilets using vacuum sewers and transported directly to an up-flow anaerobic sludge blanket septic tank (UASB-ST) for biogas production. The digestate is subject to struvite precipitation for P recovery and ammonia stripping for the recovery of ammonium sulphate both of which are used in agricultural fields. The digested sludge is composted to produce soil (75%) or applied to agricultural fields directly (25%) (Kjerstadius et al., 2017).

The GW is separately collected through a low-pressure sewer and treated in an activated sludge (AS) unit. The excess sludge is directed to the UASB-ST along with the BW. Furthermore, thermal energy is recovered using heat pumps from the GW effluent from the AS unit.

The effluent of the AS unit is treated in a post-precipitation unit before being discharged into the ocean (Kjerstadius et al., 2017). According to the case-study owners, the TW may be used for irrigation of urban farms for food production in the future. Therefore, it has been assumed here that the same quantity of TW as in the Corleone case-study is used for irrigation of these farms. Further, the case-studies also differ in the proportions of particle-bounde pollutants because of differences in temperatures and sewers.

5.4. Results

The water treatment material efficiency of Corleone (735.8 m^3/kg-linear) is much higher than that of Helsingborg (1.4 m^3/kg-linear). Helsingborg uses resource-intensive treatment processes such as struvite precipitation requiring magnesium, and citric acid and ammonia stripping requiring sodium hydroxide, sulphuric acid, and citric acid. These processes require the addition of industrial chemicals that are difficult to recycle leading to a largely linear material flow. Corleone, in contrast, makes use of lower quantities of chemicals relying more on producing TW for irrigation reuse and not on nutrient recovery processes as in the case of Helsingborg.

The use of precipitation chemicals such as magnesium chloride (for struvite precipitation) in Helsingborg adds to the chemical use of the treatment plant. But, this chemical gets recovered as part of the struvite crystals and can be recycled in agriculture. However, the bigger problems are the chemicals used for pH control, such as NaOH, which are manufactured in industries and are difficult to recycle and damage the circularity of the WWTP considerably. NaOH is used here to control the pH of the ammonia stripper and citric acid for cleaning the precipitation equipment. A high chemical use intensity of precipitation-based nutrient recovery technology has also been mentioned by Ye et al. (2020) and Sakthivel et al. (2012) among others. The chemicals used for pH control and/or cleaning are manufactured in industries and are very difficult to recycle and therefore reduce the circularity and consequently the material efficiency of the WWTP. More research is recommended into nutrient recovery technologies that require a lower chemical input such as electrochemical processes (Perera et al., 2019).

The energy efficiency of the Corleone WWTP (4.6 m^3/kWh) is also significantly higher compared to Helsingborg (0.2 m^3/kWh). The main reason for this is the energy-intensive heat pumps used by Helsingborg to recover thermal energy. These heat pumps are responsible for about 75% of the total energy used. However, this pays off in a high energy self-sufficiency value discussed later. Heat pumps are devices that can transfer thermal energy from a low-grade source (such as wastewater) to a working fluid and then raise its thermal energy content using mechanical energy (Culha et al., 2015). More energy-efficient designs for heat pumps need to be researched (Chae and Ren, 2016) but the inclusion of heat energy recovery using heat pumps is capable of supporting energy-positive WWTPs as noted by Barroso Soares (2017) and confirmed by this study.

In terms of economic efficiency, a similar relationship is found where Corleone (6.5 m^3/€) performs substantially better than Helsingborg (0.2 m^3/€). It is important to note that all costs were normalized using purchasing power parity (PPP). This result was expected as Corleone is an existing WWTP and the only capital costs it incurs are for the repair of the UF unit and the infrastructure required to connect the WWTP to a storage tank meant for irrigation water. The Helsingborg WWTP was constructed recently and thus involves green-field costs. Yet, this comparison is important for the consideration of constructing decentralized/source separation systems in the future to replace old conventional WWTPs. Since most of the existing WWTPs are of the conventional centralized type, the capital costs associated with them only pertain to the repair of the existing infrastructure. On the other hand, to construct DSS WWTPs, new infrastructure needs to be installed which adds up to significant costs.

With reference to the food production sector, Helsingborg performs better overall than Corleone. With regards to the freshwater efficiency, the Helsingborg WWTP (1.7 m^2/m^3-linear) is a better option than the Corleone WWTP (0.8 m^2/m^3-linear). This implies that nearly twice the land area can be irrigated using a unit linear flow of freshwater in the Helsingborg case than in Corleone. This is because Corleone's arid climate demands a higher evapotranspiration of a crop than in Helsingborg leading to a higher irrigation water demand for a unit area of food production. Helsingborg currently doesn't use any TW for irrigation and all of the effluent is discharged into the ocean. The Corleone WWTP only has a limited capacity of 25 m^3/h to provide TW for irrigation after ultrafiltration. Therefore, the irrigation water required for food production in Corleone is much larger than what this WWTP can currently provide leading to a low freshwater efficiency. This observation supports the view that TW reuse for irrigation is much more important in arid and semi-arid regions, such as Corleone, which are prone to water scarcity (Ofori et al., 2021) and have high evapotranspiration requirements (Liu et al., 2023). Especially in Corleone, the majority of the TW is assumed to be discharged into the Eleuterio River which has a relatively small flow rate. Thus, a large quantity of the river flow is required to dilute the nutrients and organic matter present in the WWTP effluent as revealed by the negative freshwater restoration value. Therefore, increasing the TW reuse for irrigation is strongly suggested for Corleone.

The nutrient efficiency for Helsingborg (80.4 m^2/kg-linear) is slightly better than that of Corleone (75.0 m^2/kg-linear) but the difference is not substantial. However, this was based on the assumption that the nutrients present in the TW used for irrigation will be taken up by the crops with the same efficiency as the zeolite and ammonium sulphate fertilizer products obtained in Helsingborg. This assumption needs further research. Additionally, the quality of the fertilizer products, in terms of lower heavy metal or organic micropollutants concentrations, are not accounted for by this indicator. For future research, integrated models to capture the quality of the recovered products must be developed and included in the nutrient efficiency calculations (Solon et al., 2019).

The biggest advantage for food production is seen in terms of the energy efficiency of irrigation with Helsingborg (152.9 m^2/kWh) performing almost 28 times better than Corleone (5.5 m^2/kWh). This is because firstly, the authors have assumed the average distance between the WWTP and the point of irrigation to be 0.1 km for the Helsingborg case and 2 km for Corleone. The Helsingborg WWTP is meant to be a decentralized one that can cater to local reuse of TW for the irrigation of urban farms for instance.

Secondly, some irrigation water is assumed to be sourced from underground. The average groundwater depth in Sweden is only 2 m (Barthel et al., 2021) which translates into a much lower pumping energy use when compared to an average of 25 m in Sicily (Morici et al., 2023). A sensitivity analysis (S5.4) was conducted for the distances that shows that even when a 5 km distance is assumed between the WWTP and the farm in Helsingborg, it performs better with 21.7 m^2/kWh compared to Corleone at 5.5 m^2/kWh. The reason behind this is the lower energy required to pump groundwater in Helsingborg. The food production energy efficiency can significantly benefit from the close-distance reuse of TW for irrigation. Thus DSS does help to reduce the energy use of transporting irrigation water and thereby favour local water reuse (Capodaglio, 2017). Therefore, to maximize the benefit of TW reuse for irrigation over freshwater, reducing the distance between the WWTP and the agricultural fields is crucial and decentralization can help with this. The water treatment and food production efficiency results of the two case-studies are shown in Figure 5.4.

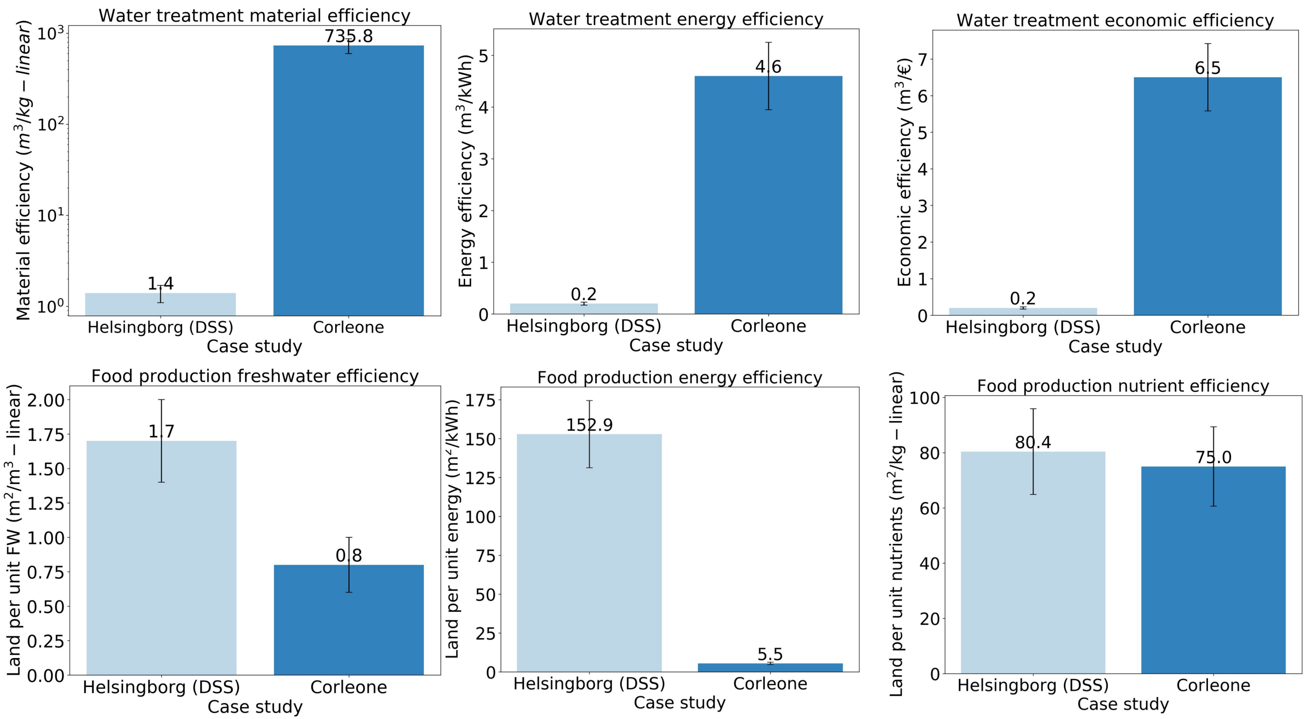

Figure 5.4: The various efficiencies for the water treatment and the food production processes. (a), (b), and (c) show that Corleone performs better than Helsingborg in terms of the water treatment efficiencies. (d), (e), and (f) show that Helsingborg is the preferred choice with regards to the food production efficiencies.

Comparing the circularities, it is evident that Helsingborg's use of chemical-intensive treatment technologies leads to a very low mixed resource circularity (1%) compared to Corleone's (40%). Corleone also performs better (84%) in comparison to Helsingborg (65%) in terms of biotic resource circularity. This is because in Helsingborg, the BW and excess sludge are digested to produce energy from biogas, and therefore the material resources are lost. On the contrary, in Corleone, the excess sludge is composted and thus, the material resources are retained. The biotic material circularity of a WWTP suffers if the COD is converted into biogas and used for energy production. This need not be a negative thing as biogas is a renewable energy source and the use of sludge products for soil application may not be suited in all cases. Therefore, in such cases, maximizing the circularity is not so important given that biogas production is quite valuable for sustainability.

The water circularities of both cases are the same (98%) because the assumption is that only 5% of the wastewater volume is lost during conveyance and treatment. Helsingborg however shows a much better performance for nutrient circularity (95%) compared to Corleone (58%). This is consistent with the fact that at Helsingborg, nutrients are recovered using struvite precipitation and ammonia stripping. In the Corleone case-study, nutrients are not recovered separately. Yet the nutrient circularity of Corleone is not negligible. Some of the nutrients present in the excess sludge become part of the compost and the nutrients present in the TW get used in agriculture with the irrigation water.

For the food production process, the water circularity of both cases is low because only a small percentage of the irrigation water requirement to produce the Wheat crop for a population of 12000 is met by TW. Both cases largely depend on groundwater. The food production water and nutrient circularities can be improved by using the recovered TW and nutrients. However, in the cases studied here, only a small percentage of the total TW was assumed to be used for irrigation thereby limiting the water circularity. This assumption was because currently TW irrigation is not practised at either location but it is a part of the plan.

The food production nutrient circularity of Helsingborg is slightly better (35%) compared to Corleone (30%). This is mainly because the quantity of nutrients recovered from the WWTP is much higher for Helsingborg than for Corleone and thus more of the agricultural fertilizer need can be met with the recovered nutrients. The circularity assessment results of the two case-studies are shown in Figure 5.5.

When it comes to the three nature reciprocity indicators, Helsingborg is found to be the better option. No freshwater restoration is achieved by Helsingborg because some of the TW is used for irrigation and the rest is discharged into the ocean. This is not necessarily a negative point as long as the effluent quality does not disturb the ocean ecosystem. Contrary to this, the Corleone WWTP discharges its TW into the Eleuterio River. However,

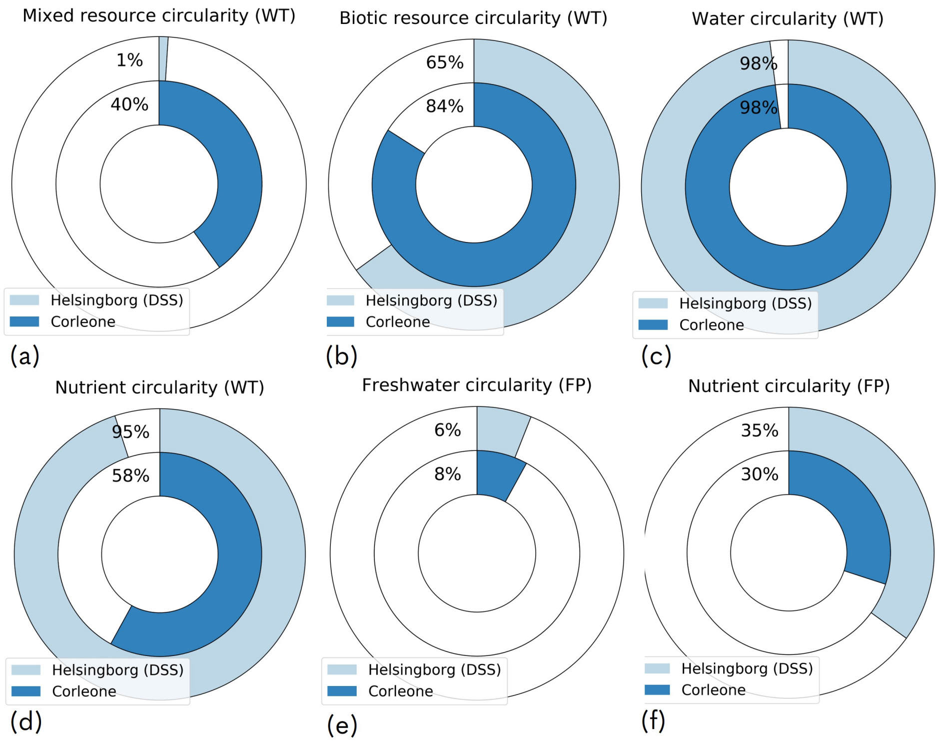

Figure 5.5: The various circularity values compared for Helsingborg and Corleone. Corleone performs better in terms of the mixed and the biotic resource category. For the food production, Corleone performs slightly better for water circularity but Helsingborg has a higher nutrient circularity.

the FR value is -8.1×10^7 m^3/y implying that the TW requires a large quantity of water for its dilution. Although dilution of the TW discharge does not consume the river water, the river water quality can suffer through the non-consumptive use of it. The streamflow of Eleuterio is simply insufficient to provide this dilution capacity therefore, reuse of the TW is strongly recommended instead of the discharge to avoid a degradation in the river water quality. Arid regions like Corleone are expected to have even lower stream flows in the future due to climate change and therefore, reuse of the TW for irrigation or other purposes is strongly recommended.

The biomass assimilation of the nutrients is higher (2.9×10^4 kg/y) for Helsingborg than Corleone (1.1×10^4 kg/y) as expected because the nutrients are recovered as ammonium sulphate and struvite to be used in agriculture. In the Corleone case-study, the nutrients are not recovered by a dedicated recovery process but, are used in agriculture through the application of TW and the sludge compost. Consequently, the nutrient recovery efficiency of Corleone (43% for N and 38% for P) is lower than that of Helsingborg (78% for N and 98% for P). Further study into incorporating a nutrient recovery process such as struvite precipitation, ammonia stripping, or any other effective process is recommended for Corleone to improve the nutrient recovery efficiency.

The soil organic matter sequestration of Helsingborg (20.8×10^4 kg/y) is also higher than Corleone's (9.02×10^4 kg/y) mainly because the volatile solid component of the sludge after anaerobic digestion (35% in Helsingborg's case) is lower than that of the compost (60% in Corleone's case). The lower VS content leads to more of the organic matter present in the sludge being sequestered into the soil.

The Helsingborg WWTP achieves an energy self-sufficiency of over 200% implying that more than twice the energy (electricity+heat) spent on the treatment is recovered. This was despite the higher energy consumption of the DSS WWTP when compared to the conventional one. Including thermal energy recovery using heat pumps contributes to the high self-sufficiency of Helsingborg. A high thermal energy recovery is favoured by the GW separation as this stream contains most of the heat energy (Larsen, 2015). The thermal energy that can be recovered from DSS WWTPs is estimated to be between 477 kWh/capita/year and 840 kWh/capita/year (Kjerstadius et al., 2016). Thus, there is a large variability in this. In any case, the recovery of thermal energy with source separation treatment is very promising for energy-positive WWTPs. In contrast, energy is not recovered from the Corleone WWTP and it relies entirely on an external energy supply. The nature reciprocity and the energy self-sufficiency results are shown in Figure 5.6.

5.5. Discussion

5.5.1. DSS vs conventional centralized treatment

Using the WFE framework for the comparison revealed the different pros and cons of the two approaches. Firstly, the material efficiency of water treatment is strongly dependent on the type of treatment technology used irrespective of the decentralization scale. This means that the use of chemical-intensive treatment will negatively affect the material efficiency of the treatment and these should be avoided.

Secondly, most of the advantages of the DSS treatment result from the source separation of BW and GW. These advantages include a high efficiency of biogas recovery, a high thermal energy recovery, and higher nutrient recovery. The main advantage of decentralization is the reduction of energy use for the transport of irrigation water. However, this advantage may not be significant in regions like Helsingborg with a shallow water table depth. Therefore, the advantage of decentralized TW reuse is most prominent in arid and semi-arid regions with seasonal stream flows and low water table depths. Consequently, while planning the future infras-

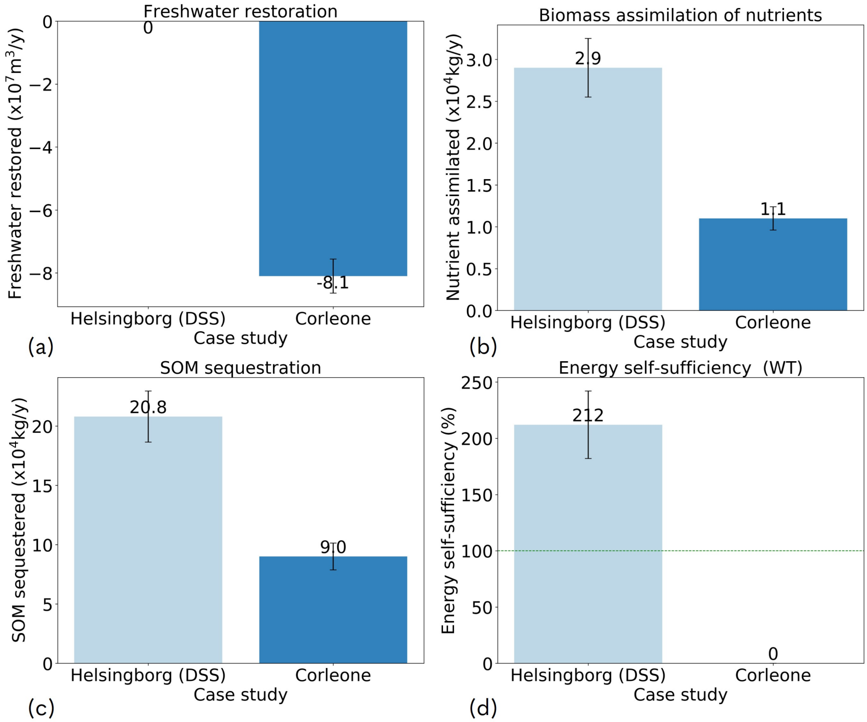

Figure 5.6: The nature reciprocity and the energy self-sufficiency compared for the Helsingborg and Corleone case-studies. Helsingborg performs better on the four criteria.

tructure, source separation should be implemented before decentralization because it may offer more benefits. However, in arid and semi-arid regions, decentralization may be given equal importance.

Thirdly, it is interesting to note that maximizing the circularity need not always be desirable subject to the context. If the excess sludge is digested for energy recovery, some biotic material may be lost causing a lower circularity. This is the case in Helsingborg which has a biotic resource circularity of 65% compared to the 84% in Corleone. However, the recovery of renewable biogas energy is desirable despite reducing the WWTP's circularity. The decision-makers need to consider whether they want to maximize the circularity or recover renewable energy.

5.5.2. Advantages of the WFE framework

The proposed WFE framework allows us to use the functional units and indicators that are of direct relevance to each sector thereby facilitating the communication of their benefits. The framework makes it possible to evaluate the effect of resource recovery directly on the water treatment efficiency. The efficiency assessment of the water treatment sector helps to compare the case-studies on their performance solely from the perspective of treating the wastewater to the relevant effluent standards. Additionally, the framework can explicitly quantify the benefits to the food production sector using indicators that are directly relevant to it. These benefits were shown here in terms of higher nutrient, and freshwater circularity and efficiency in the production of the wheat crop and also in the form of a higher energy efficiency of food production. A lack of clear communication within the nexus results in a lack of integrated planning and management of the resources and an inclusive tool is required to bridge the communication gap (Mohtar and Daher, 2016). The clear communication of the benefits of resource recovery to the water treatment and food production sectors is the first advantage of this WFE framework that will likely lead to a more holistic and integrated resource management.

This framework can account for the relevant local climate and geographical factors. The food production efficiency assessment is done for the freshwater, the nutrients and the energy flows. The agricultural conditions of different regions demand varying quantities of water based primarily on the evapotranspiration (ET) needs of a crop. TW may be used to grow a wide variety of crops. Standardizing the crop to the common wheat offers a proxy way of judging the efficiency of the food production sector irrespective of the crop type and aids the comparison of different real-life case-studies while still retaining the local climate factors in the form of the evapotranspiration value. Therefore, a second advantage of the framework is that it can help account for the agricultural water requirements subject to the local climate.

Furthermore, this is the first WFE framework that incorporates nature reciprocity indicators. This ensures that the sectors take responsibility for making a positive impact on the natural environment. The FR nature reciprocity indicator can also help to evaluate if discharging the TW is causing too much pressure on the natural streams as shown in the Corleone case-study where the FR value is negative. A negative FR value informs the decision-maker of the insufficient dilution capacity of the natural stream.

A lack of acceptance of TW reuse for irrigation from a certain percentage of the population has been discussed by Verhoest et al. (2022) and Saliba et al. (2018). Verhoest et al. (2022) pointed to the crucial need to make people aware of how their decision to consume TW-irrigated crops can protect the natural environment. Showing that discharge of the TW could be having a net negative effect on the natural streams can contribute to this and can potentially lead to higher acceptance which is the third advantage of the framework.

5.5.3. Limitations and future outlook

The WFE framework does not account for the revenue generated by a WWTP in return for providing the recovered resources due to the lack of data. This needs to be considered in the future for a more accurate economic efficiency assessment. Furthermore, the benefit assessment of discharging TW into the ocean or a lake is not captured by this framework since only the FR benefits to a stream can be assessed. In the future, methods to assess the other benefits should be developed.

In addition to above, the circularity assessment in the WFE framework is focused on mixed materials, biotic resources, nutrients, and water. While these resources are mostly targeted for recovery, a few other resources such as metals are also sometimes recovered. For a more comprehensive assessment framework, methods to assess the circularities of other resources should also be developed.

Another limitation of this study is that the authors have not accounted for the quality of the recovered products. The chemical- and energy- intensive treatment processes of Helsingborg that were damaging from the energy efficiency and circularity perspectives are important because they also lead to a higher quality of the recovered products. As an example, the recovered nutrients contain lower heavy metal and PFAS concentrations. Also, the higher quality of water being used for irrigation is not captured by this framework. For future work, indicators need to be developed that can capture the effect of improved quality of the recovered resources.

Additionally, the economic efficiency assessment in this chapter included a comparison between the capex of upgrading an existing centralized WWTP with that of the green-field costs of a DSS treatment plant. This naturally favours the existing infrastructure. The DSS treatment plants can be seen as a replacement for the conventional centralized ones at the end of their service period. Then, the choice would be between constructing a new conventional or a DSS WWTP. Therefore, a life cycle cost comparison between the two is recommended for future work.

Lastly, the WFE framework lacks the assessment of social factors such as public acceptance because this is not the expertise of the authors. It is recommended to include the social dimension in the framework in the future.

5.6. Conclusions

The chapter provides a novel WFE framework to compare a conventional centralized and a DSS wastewater treatment approach to resource recovery. The framework with its assessment methods achieves the following:

- It covers multiple dimensions, such as economic performance, nature reciprocity, efficiency of water treatment and food production, energy self-sufficiency of the water treatment process, and circularity.

- It takes into account the local climate and the agricultural conditions by including factors such as the agricultural land use per capita and the evapotranspiration needs of crops.

- It contains indicators specific to the water treatment and food production sectors that make it easier to communicate the benefits of resource recovery.

- It can potentially help to increase the acceptability of treated wastewater reuse for irrigation by explicitly showing the positive/negative effect of treated wastewater discharge into a stream.

These make the new WFE assessment framework more integrated and holistic compared to the existing frameworks.

The new framework was applied to two case-studies, one consisting of a decentralized source separation treatment (Helsingborg) and the other one being a conventional centralized treatment (Corleone). The related assessments of two different approaches resulted in the following observations:

- The Helsingborg approach to heat and electricity recovery leads to an energy-positive water treatment. However, the construction of the new infrastructure leads to a lower economic efficiency. Additionally, the chemical- and energy- intensive processes reduce its material and energy efficiencies but also lead to better quality recovered resources and new methods to quantify the resource quality need to be developed. The food production becomes substantially more efficient and circular by the Helsingborg approach.

- In Corleone, the centralized infrastructure is already present and therefore this remains the more economically efficient approach atleast for the duration of the infrastructure life. Due to a low water table in Corleone, replacing groundwater with treated wastewater for irrigation is strongly recommended. This would also reduce the pressure on the local river where the effluent is discharged.

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6

Conclusions, contributions, and future work recommendations

6.1. Conclusions

In this thesis, new methods were developed to assess the circularity, efficiency, and nature reciprocity for the water treatment sector. To start with, three strengths and weaknesses of the LCSA framework were expounded to suggest future research directions in Chapter 2.

In Chapter 3 a novel circularity assessment method was developed for the resource recovery solutions. First, the restorative, regenerative, and linear flows were redefined in a manner that fits the technical cycle and the biogeochemical resources present in the water sector. Thereafter, equations for the circularity calculation were presented by modifying the existing material circularity indicator method.

Chapter 4 emphasized the need for assessing the potential benefits of resource recovery on the natural environment. An innovative method was introduced to assess three benefits: Freshwater restoration, biomass assimilation of nutrients, and soil organic matter sequestration.

In Chapter 5, the lack of a water-food-energy framework to compare the decentralized source separation and the conventional centralized approaches to wastewater treatment and resource recovery was identified. Hence a new framework was presented containing indicators relevant to the three sectors that can help assess real-life cases in an integrated and holistic manner. This framework can simplify the communication between the sectors, leading to better coordination.

Four research gaps identified via literature review led to the formulation of corresponding research questions (see Chapter 1 for details). Thesis conclusions are presented below as answers to these questions.

RQ1:- What are the strengths and weaknesses of life cycle sustainability assessment in the context of resource recovery solutions linked to wastewater treatment plants?

The strengths of the LCSA framework in the context of the water treatment sector, briefly discussed in Chapter 2, include the following. Firstly, LCSA avoids burden-shifting between different environmental issues and/or life cycle phases, by including a large number of environmental impact categories and by covering the entire life cycle of the resource recovery and the resource application processes. Secondly, it includes the environmental, economic, and social dimensions of resource recovery solutions. This helps in a holistic assessment by including indicators from the three dimensions. Thirdly, LCA is an environmental damage evaluation method used as part of LCSA. This method can inform users about the environmental performance of a resource recovery solution and help reduce its negative environmental damages (improve its eco-efficiency).

The weaknesses of LCSA elucidated in Chapter 2 are as follows. LCSA is entirely focused on the assessment of environmental damages. However, the recovered resources can be also used to actively benefit the natural environment but LCSA lacks the means to assess these benefits. Furthermore, the frequent use of compensatory aggregation methods to combine the indicators of the three dimensions is an ontological weakness of the framework. Unrestricted compensations between the environmental and the economic dimensions is inconsistent with the fact that the economy depends upon the natural environment. Treating economic development and environmental damage as completely substitutable factors could lead to continued environmental damages which would also limit future economic development. Lastly, the LCA indicators without the context of current emissions/resource stocks and some threshold value can potentially underestimate the urgency to prevent or reverse an environmental damage. These factors are necessary to include in an LCA to accurately judge how sustainable a resource recovery solution really is.

RQ2:- How can a method be developed to accurately assess the circularity of the biogeochemical resources present in the water treatment plants?

A new method that can assess the circularity of the biogeochemical resources was developed by redefining the restorative, regenerative, and linear flows in a way that fits these resources.

Restorative flow is a flow that recovers a resource for direct human use (e.g., recovery of the struvite out of wastewater through precipitation).

Regenerative flow is a flow that returns a resource to the state in which it was originally appropriated from nature for human use. This is to promote the self-renewal and ecosystem-sustaining capacity of biogeochemical cycles in response to overexploitation (e.g., releasing reactive nitrogen as N_2 into the atmosphere to close the nitrogen cycle).

Linear flow is a flow that is obtained from virgin sources and/or discarded in a form different from how the resource was originally obtained for human use (e.g., returning water obtained from a river as water vapour to the atmosphere).

Using above definitions and the modified MCI method presented in Chapter 3 enables to assess the circularity of the resource recovery solutions more accurately. The new method also accounts for the environmentally functional flows (e.g., groundwater infiltration of irrigation water) and environmental losses (e.g., evapotranspiration of irrigation water) of the biogeochemical resources. This ensures that the resource flows associated with the resource recovery solutions are correctly identified and considered, thereby leading to a more accurate circularity assessment.

Fertigation using treated wastewater can improve the water and nutrient circularities compared to using freshwater and industrial fertilizers. However, careful attention needs to be drawn to the evapotranspiration, and underground infiltration of the water. Furthermore, agricultural drainage must be collected to maximize nutrient and water circularities.

RQ3:- What are the potential nature benefits from the application of the recovered resources and how can they be maximized?

The three potential nature benefits from the resource recovered from the water treatment sector are freshwater restoration, biomass assimilation of nutrients, and soil organic matter sequestration. These can be assessed using the nature reciprocity method presented in Chapter 4.

The freshwater restored can be maximized by increasing the recovery of nutrients and organic matter and discharging the treated wastewater into a high flow-rate stream. The biomass assimilation of nutrients can be augmented by using recovery processes that lead to a high recovery efficiency but also produce a nutrient product that can be efficiently taken up by the plants. An advantage of WWTPs is that the nutrient products are usually of a slow-release type that favours a high uptake efficiency. Lastly, sludge products with low volatile solid components can be used to provide stable sequestration of organic matter into the soil.

The reciprocity assessment is complementary to the environmental damage assessment. Assessing both the aspects together may reveal new resource recovery and application pathways and lead to better decision outcomes.

RQ4:- How can the conventional centralized and the decentralized source separation approaches to wastewater treatment and resource recovery be compared from a water-food-energy nexus perspective in an integrated and holistic manner?

The comparison is possible by using a holistic framework including multiple dimensions including the economic performance, circularity, efficiency, and nature reciprocity and by using the indicators relevant to the three sectors. The use of sector-specific indicators can ensure that the benefits of the resource recovery are clearly communicated to the different stakeholders. This can in-turn improve the inter-sectoral coordination.

Treating wastewater through a decentralized source separation (DSS) approach may require a higher chemical, energy, and economic input, especially when the quality of the recovered resources is high. However, a DSS approach may ensure a more circular and efficient production of food and may ensure an energy positive wastewater treatment.

6.2. Scientific contributions

A new method for the circularity assessment of resource recovery solutions is presented in Chapter 3. This method is based on important modifications made to the existing MCI method. In the new method circularity assessment is combined with a dynamic model of resource flows which makes the assessment more accurate. The new method also accounts for the complexities of the biogeochemical resource flows. The proposed method can be used to conduct assessment studies related to sectors other than the water treatment sector. The definitions presented as part of this method are applicable to the biogeochemical as well as the technical cycle resources. Further, the method was used to assess the circularity improvements when switching from freshwater irrigation and industrial fertilizers to treated wastewater (TW) fertigation. The case-study results contribute to the literature discussing the benefits of using TW for irrigation.

The new nature reciprocity method, presented in Chapter 4, provides a new vision for the science of sustainability. In the current sustainability literature, it assumed that humans can only have negative impacts on nature and all that needs to be done for sustainability is to reduce these negative impacts. However, the innovative nature reciprocity method highlights how resource recovery solutions can positively impact the natural environment and offers ways to assess these impacts. This may also be true of sectors other than water treatment and the method has a wide applicability. This assessment will ensure that future research in sustainability also considers the potential positive impacts on the natural environment. Additionally, the newly developed method was applied to a real-life WWTP that primarily relies on physio-chemical processes. The study also described the mass flows of water, nutrients, and organic matter, which could be useful for future assessments.

Chapter 5 provides a new WFE framework to aid in the transition towards decentralized source separation. There was a lack of an WFE framework to assess the conventional centralized and decentralized source separation approaches to wastewater treatment and resource recovery. Since the water treatment sector is gradually adopting decentralization and source separation, the effects of this transition on the other sectors needed to be studied. The novel framework may be used whenever the two approaches need to be compared considering locally relevant factors. The framework was applied to two case-studies with real-life data. These results contribute to the WFE framework literature as well as the assessment of decentralized source separation and can be used to discuss the advantages and disadvantages of other cases where a similar transition is planned.

6.3. Societal contributions

The developed methods can help the professionals and decision-makers dealing with water and wastewater treatment to better assess and compare multiple resource recovery options pertaining to the resources to be recovered and the technologies to be used for it. These methods may particularly be used as aids to discussions in the planning phase. This would ensure a systematic planning of a resource recovery solution that considers crucial factors such as efficiency, circularity, and nature reciprocity.

The circularity assessment method presented in Chapter 3 may be used to accurately evaluate the resource recovery solutions by helping the decision-makers to correctly classify the biogeochemical resource flows as linear, restorative, or regenerative. The method can be used based on simple mass flow data related to the water or wastewater treatment plants and the process wherein the recovered resources are used (e.g., irrigation). It can account for the local climate and soil conditions and also the crop type to be irrigated. This is particularly relevant for the reuse of treated wastewater and nutrients in the agricultural sector. However, other sectors can also make use of this method as long as an accurate resource flow model is available. In this method, a flow that returns a biogeochemical resource to the natural environment in an acceptable form is considered a regenerative flow. Thus, using this method can encourage the decision-makers to also think about the way in which a resource is returned to the natural environment. For example, when conducting the assessment of using treated wastewater for irrigation, the method requires of the decision-maker to consider the water quality of the run-off and the underground infiltration too hence provides an accurate and a systematic way to assess the analysed resource recovery solution.

The nature reciprocity method presented in Chapter 4 can serve the decision-makers in several different ways. It allows them to consider the positive impact on nature through the resource recovery solutions. This knowledge will enable the decision-makers to properly weigh the positive and the negative effects of a resource recovery solution and can lead to a different discussion and outcomes than if only the negative impacts were assessed. Moreover, this method allows them to consider the pressing environmental issues in the local environment and include these in their decision-making process. For example, if the local soil is severely depleted in organic matter, then its replenishment using sludge products can be included and prioritized in the discussion using the nature reciprocity assessment. Similarly, the method can also reveal if the effluent discharge into a local stream is having a restoring or a degrading effect. WWTPs may face challenges convincing the local authorities and stakeholders to reuse the treated wastewater. The knowledge of the negative impact that the effluent discharge may be having on the streams can lend support to the reuse option for the treated wastewater. It could also drive the discussion on what type of nutrient recovery technology to use and what kind of nutrient products are most desirable for maximizing the nutrient uptake for the local agricultural conditions.

The WFE framework presented in Chapter 5 represents a practical way to evaluate and compare the conventional centralized and the decentralized source separation approaches. The methodology of this framework is generic and reproducible hence can be applied to case-studies anywhere in the world. The framework makes use of data that are easily accessible to the decision-makers. It also accounts for relevant local conditions including the distances between farms and WWTPs, the agricultural land use per-capita, and evapotranspiration needs of the crop in the local climate. It includes indicators that are relevant to the water treatment and the food production sectors and this is likely to help in clear inter-sectoral communication. It can thus help the stakeholders from the two sectors with better coordination and consequently better management of the resources. The application of this framework on two semi-hypothetical case-studies revealed that many of the benefits of DSS likely stem from separating the grey and black water streams. Hence, it is advisable to prioritize source separation as the initial step in developing future water treatment infrastructure.

6.4. Future work recommendations

6.4.1. Life cycle sustainability assessment

It is imperative to include the current emissions/resource stocks and some environmental thresholds (such as reactive nitrogen and atmospheric CO_2 concentration limits) when communicating about sustainability. While some work has been done lately in this direction, scaling the planetary boundaries (such as climate change) down to a magnitude relevant to a water treatment plant has received little attention. An advantage in the case of a water treatment plant is that we know the people equivalents served. This could be a starting point to assign certain thresholds or limits to the emissions or resource consumption of a certain treatment plant per-capita. This would need the consideration of many socio-economic and geographical factors too. This could be achieved in several ways including bench-marking the treatment plants located in similar socio-economic and geographical situations. Thus more research is needed to scale down the planetary boundaries and include them when communicating LCA results related to water treatment plants.

The non-compensatory relationship between natural and economic capitals was discussed in this thesis. Several studies make use of MCDA methods that freely compensate between the economic and the environmental performance indicators. In-fact, it is this indiscriminate utilization of the natural capital for economic development that has globally caused environmental degradation. Therefore, such an approach is not entirely consistent with the concept of sustainability. Methods that limit the compensation between the two criteria or completely eliminate it are more appropriate for application to the sustainability assessment of the water treatment sector. Non-compensatory MCDA methods are available and also recommended for sustainability assessments by several authors. However, their use to assess water treatment and resource recovery solutions remains limited. Therefore, a comparison study is recommended to see if and how the ranking of resource recovery alternatives changes when using compensatory and non-compensatory MCDA methods. In the case that the rankings change substantially, it is recommended for future LCSA applications to the water sector to utilize one of the non-compensatory methods.

Moreover, the compensatory aggregation also reveals our assumption that economic development is opposed to environmental preservation. This view has to undergo a transformation. For real sustainability, we must aspire for an economic developmental model that also has a net benefit to the natural environment. Therefore, new economic developmental models must be studied that can actively benefit the natural environment. Agroforestry is one such model used in agriculture that can serve as an example for the water treatment sector.

6.4.2. Resource flow models

The new circularity assessment method (Chapter 3) relies on the modeling of the flow of resources through the WWTPs and the processes where the recovered resources are used. The utility of this method is strongly dependent upon the accuracy of the resource flow model used. Several field-specific models exist but these may not be easily accessible to the decision-makers. Simpler models need to be developed for the different WWTP configurations and the most commonly used reuse processes such as agricultural fertigation, anaerobic digestion, and composting. These models should be able to characterize features such as the local climate, the agricultural practices, the hydrological and the soil characteristics. Ideally, such a model can be coupled to a spreadsheet-based circularity assessment tool. This would serve the decision-maker in making an accurate assessment of the circularity of the agricultural reuse of treated wastewater and nutrients.

6.4.3. Nutrient uptake efficiencies

An innovative nature reciprocity assessment was presented in Chapter 4 but more work is required to build upon this idea. An accurate estimate of the nutrient uptake efficiency (NUE) of the different recovered nutrient products is very important to calculate the corresponding nature benefit. The NUE depends on the agricultural practices, the soil characteristics, and several other factors. Models need to be developed that accurately predict the NUE of the different nutrient products such as struvite and vivianite. Additionally, pot experiments may also be used to estimate the NUE for specific conditions that resemble the actual farm conditions.

6.4.4. Recovered resource price and quality

The data on the revenue from selling the recovered products from the water treatment sector are not easily accessible. This is especially the case in the planning phase when the assessment methods presented in this thesis are likely to be used. This may be due to a lack of consistent information about the quality of the recovered products, a lack of a market, or competition with the existing products distributed by the well-established companies. There is a need for studies that report the revenue generated from the sale of the recovered resources. Another option is to develop accurate methods to estimate the price of a recovered product based on its quality. This links to another topic that needs research and is presented below.

The quality of the products recovered from a water or wastewater treatment plant can vary depending on the water characteristics and the utilized recovery process. Currently, methods to quantify the quality of these products in a rigorous way are not available. A recovery process may achieve better quality products at the cost of higher energy and chemicals consumption but the higher quality is not captured by the methods present in this thesis, thereby disadvantaging such processes. Therefore, methods to describe the quality of the products recovered from the water treatment sector need to be developed.

6.4.5. Water-food-energy nexus

In Chapter 5, an existing conventional centralized WWTP was compared to a newly built decentralized source separation (DSS) WWTP. This meant that the capital costs associated with the latter were substantially higher. In the future, as the existing WWTPs reach the end of their operation period, new WWTPs will need to be constructed. To decide which of the two approaches is suitable for the future, more studies are needed that compare the life cycle costs of constructing conventional WWTPs and DSS ones.

6.4.6. Further validation of the new methods

The new methods (Chapters 2,3, and 4) and the novel WFE framework (Chapter 5) presented in this thesis were each tested on one or two case-studies. These methods need to be validated on additional, more complex case-studies located across different geographical regions. Therefore, the applications of these methods and the WFE framework on other resource recovery solutions are recommended.

A

Appendix- Life cycle assessment

A.1. Goal definition

The LCA will reveal environmental impacts associated with the production of the bio-composite material and identify hotspots. The study is targeted towards the bio-composite manufacturing company, water treatment facilities and academics. This LCA is being conducted primarily to demonstrate the limitations of the LCSA framework and hence the outcomes should be seen as indicative. The actual impacts calculated can serve as basis for a more detailed LCA in the future.

A.2. Scope definition

A.2.1. Functional unit and reference flow

There is a large number of processes involved in producing any product. The scope of an LCA has to be limited to key processes and this is why a system boundary is defined. Here, the system begins from raw material extraction. For the bio-composite, two raw materials are extracted from the urban water cycle: Calcite from drinking water treatment and water reeds from canal banks. Calcite is crushed into small particles and the reeds cut into 3-6mm fibres. The two are mixed with unsaturated polyester resin and other chemical additives to form a Bulk Moulding Compound (BMC). The BMC is hot-pressed to form boards which are used as canal bank protection. They are assumed to have a life time of 25 years after which the boards are ground into small particles and reused in the bio-composite manufacturing. Thus, this is a cradle to cradle LCA. The flowchart is shown in Figure 6.4.

A.2.2. Data relevance

Primary data on the manufacturing process is obtained from the manufacturer. Thus, the foreground system is modelled within the same geographical boundary as in reality. For electricity generation, depending on the location of the unit process, either the energy mix of the Netherlands or that of Germany is used.

A.2.3. Impact categories

According to ISO 14044, the selected impact categories must cover a comprehensive set of issues relevant to the studied product (**?**). There is no consensus on the choice of impact categories pertinent for bio-composites (Vidal et al., 2009). Therefore, in order to cover the widest range of impacts, the ReCiPe 2016 method is used which covers seventeen environmental issues. A first run reveals the most relevant categories which become the focus of further analysis.

A.2.4. Multi-functionality

Water softening is a multi-functional process in the LCA, producing two products namely, softened water and calcite. Breaking down the process into two sub-process is first considered. But dividing the process into one that creates calcite and another that softens water does not seem possible. System expansion is the next consideration. The softening process can be attributed with an avoided burden of producing calcite industrially. However, the quality of recovered calcite is not known with certainty and thus, whether it can replace industrial calcite for other applications is highly uncertain. Hence, price-based allocation is used.

A.3. Life cycle inventory

A.3.1. Foreground processes

The manufacturing process contains two main ingredients which are calcite from drinking water treatment and water reeds. Calcite is obtained from water softening which is a multi-functional process. Price-based allocation is used to attribute burdens of softening to calcite production. Inputs to the softening process which are caustic soda and electricity were obtained through Waternet. Reeds are assumed to be collected from canal sides and transported to a facility where they are milled into 3-6 mm fibres. Electricity use for this is assumed to be 0.0386 kWh/kg (Cao, 2019). Some chemicals are expected to be used as additives in the manufacturing of the bio-composite. Due to absence of precise information of these additives, a general input of organic chemicals is assumed and its quantity obtained from Cao (2019). Transportation distances are estimated using Google maps. It is assumed that the reeds are collected near Amsterdam and transported to a factory located in Osnabrück, Germany (260 km). Likewise, calcite is transported from Nieuwegein (Netherlands) to Osnabrück

(240 km). At end of use phase after 25 years, the bio-composite is collected to be shredded and ground into fine powder. Depending on process scale and machinery used, energy use for mechanical grinding is between 0.1 and 4.8 MJ/kg (Buggy et al., 1995). An average energy consumption of 2.45 MJ/kg is assumed for recycling the bio-composite. This recycled material is assumed to replace 20% of the fiber input.

A.3.2. Biogenic carbon

Water-side reeds grow naturally along canal or river banks. Over their life time 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 in it 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 which adds up to 1.459 kg/m^2 (Zhou et al., 2009). 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 as shown below.

Above ground biomass (dry mass) of water reeds = 1.459 kg/m^2 (Zhou et al., 2009). Therefore, 1 kg biomass grows in 1/1.4 m^2. Net uptake of CO_2 from water reeds = 65±14 g C/m^2 Thus, per unit mass, CO_2 uptake = (65 ± 14)×1/1.4 = 46.42 ± 10 g C.

A.3.3. Economic allocation

Price of calcite = 5 €/t
Price of soft water = 0.9 €/m^3
Total economic output = 5×5252 + 0.9×6×10^7 = € 5.4×10^7
Allocated burden percentage to calcite = (5×5252)×100/54026260 = 0.048%
Allocated burden to soft water = (0.9×6×10^7)×100/5.4×10^7 = 99.95%

A.4. Life cycle impact assessment

A.4.1. Characterization

The ReCiPe 2016 characterized results for the system shown in Figure S.1 are shown in Table A.1 along with the major elementary flow responsible for the impact and the major activity (hot-spot).

Table A.1: LCIA results for the bio-composite application to canal bank protection. The table shows impact categories, their units, values, the primary responsible substance and activity from the life-cycle.

Impact category	**Unit**	**Impact**	**Major elementary flow contributor (compartment)**	**Major activity contributor**
Global warming	kg CO_2 eq.	8.77×10^{1}	CO_2, fossil (air)	82% Polyester resin production
Stratospheric ozone depletion	kg CFC11 eq.	6.70×10^{-4}	Dinitrogen monoxide (air)	98% Polyester resin production
Ionizing radiation	kBq Co-60 eq.	5.99	Radon-222 (air)	56% Polyester resin production, 27% Shredding and grinding for recycling
Ozone formation	kg NOx eq.	1.15×10^{-1}	Nitrogen oxides (air)	85% Polyester resin production
Fine particulate matter formation	kg PM2.5 eq.	9.54×10^{-2}	Sulphur dioxide (air)	89% Polyester resin production
Ozone formation, Terr. ecosystems	kg NOx eq.	1.62×10^{-1}	Nitrogen oxides (air)	85% Polyester resin production
Terrestrial acidification	kg SO_2 eq.	2.31×10^{-1}	Sulphur dioxide (air)	87% Polyester resin production
Freshwater eutrophication	kg P eq.	3.62×10^{-2}	Phosphate (water)	43% Polyester resin production, 36% Shredding and grinding for recycling
Marine eutrophication	kg N eq.	3.22×10^{-3}	Nitrate (water)	56% Polyester resin production, 28% Shredding and grinding for recycling
Terrestrial ecotoxicity	kg 1,4-DCB	2.22×10^{2}	Copper (air)	86% Polyester resin production
Freshwater ecotoxicity	kg 1,4-DCB	3.70	Copper (water)	78% Polyester resin production

Impact category	Unit	Impact	Major elementary flow contributor (compartment)	Major activity contributor
Marine ecotoxicity	kg 1,4-DCB	4.84	Copper (water)	77% Polyester resin production
Human carcinogenic ecotoxicity	kg 1,4-DCB	2.89	Chromium VI (water)	62% Polyester resin production 23% Shredding and grinding for recycling
Human non-carcinogenic ecotoxicity	kg 1,4-DCB	7.71×10^{1}	Chromium VI/Zinc (water)	68% Polyester resin production

A.4.2. Normalization

Characterized results for the various impact categories are of incomparable units. In order to represent them on a single graph and compare their relative magnitudes, normalization is performed. In this step, the impacts of a system are compared to those of a reference average, like a country or the world (Hauschild et al., 2018). The reference used here is total impact per category of the world in 2010 as part of the ReCiPe 2016 method. Figure A.1 shows normalized impact results for the bio-composite life cycle along with their uncertainty range expressed as standard deviation.

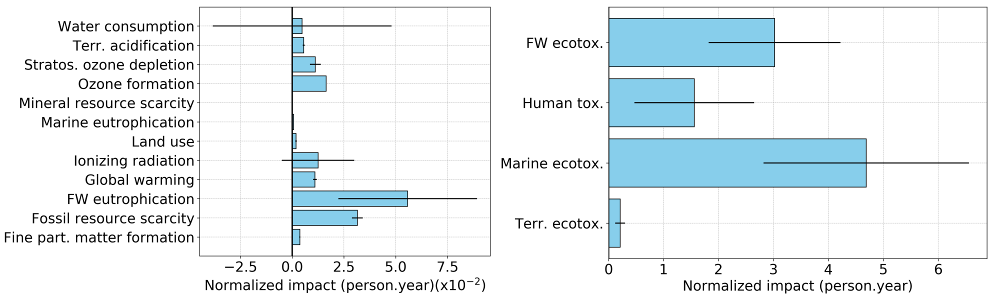

Figure A.1: Life Cycle Impact Assessment of the bio-composite material application as canal bank protection normalized to world average per-capita emission per annum. The chart on the right shows four categories that have the highest normalized impacts that are several orders of magnitude higher than the categories on the left.

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) with the last one representing a combined impact of non-carcinogenic and carcinogenic human toxicity categories. FET and 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. 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.

The impacts on the remaining thirteen categories are several orders of magnitude lower. Among these lower magnitude categories, freshwater eutrophication (FE) and fossil resource scarcity (FRC) are the two most important ones. FE damage is overwhelmingly contributed (98%) by phosphate emissions to water from coal mining for electricity used in shredding and grinding of the bio-composite. FRC is mainly caused by crude oil consumption in resin manufacturing. Also these two impacts can be traced back to the polyester resin manufacturing which is the hotspot of the bio-composite life cycle. For a complete list of impacts, their main responsible elementary flows and processes, Table A.1 may be consulted.

A.5. Uncertainty analysis

Monte Carlo analysis with 1000 iterations is used to estimate the uncertainties of impacts. The mean impact, standard deviation and coefficient of variation are presented in Table A.2. A CoV greater than 40% is seen for the categories of freshwater eutrophication, human toxicity, ionizing radiation, terrestrial ecotoxicity and water consumption. The toxicity indicators which are found to be the most important from a per-capita world average perspective also seem to have a very high variability. This high variation for toxicity categories is also seen in Niero et al. (2014) where it is primarily attributed to high uncertainty in estimating copper emissions into water. In Sleeswijk et al. (2008), the authors highlight high uncertainties in toxicity-related categories mainly due to geographical and human variability. They suggest to use the results as triggers for further investigation.

A.6. Contribution analysis

Figure A.3 shows the contribution of various life cycle stages to the overall burdens for each category. Overall, the process contributing most to all impacts is polyester resin manufacturing. Especially prominent are the ecotoxicity categories and they can be traced primarily to copper emissions into water from the resin manufacturing process. The second most important contributor to the negative environmental impacts is the recycling process. This is due to high fossil-based electricity consumption. As a result of using coal-based energy large impacts in the categories of global warming, ionizing radiation, freshwater eutrophication, marine ecotoxicity and human toxicity are seen.

Table A.2: Mean, standard deviation and coefficient of variation for the ReCiPe 2016 categories, generated with 1000 Monte Carlo iterations.

Impact category	**Mean**	**Standard deviation**	**Coefficient of variation (%)**
Fine particulate matter formation	3.73E-03	3.40E-04	9.12
Fossil resource scarcity	3.15E-02	2.63E-03	8.35
Freshwater ecotoxicity	3.06E+00	1.20E+00	39.44
Freshwater eutrophication	5.61E-02	3.35E-02	59.69
Global warming	1.10E-02	9.80E-04	8.94
Human carcinogenic toxicity	1.00E+00	1.09E+00	108.94
Human non-carcinogenic toxicity	5.27E-01	2.48E-01	47.10
Ionizing radiation	1.19E-02	1.75E-02	146.79
Land use	1.92E-03	4.49E-04	23.39
Marine ecotoxicity	4.75E+00	1.87E+00	39.36
Marine eutrophication	6.99E-04	1.06E-04	15.23
Mineral resource scarcity	1.80E-06	6.82E-07	37.98
Ozone formation, Human health	7.35E-03	6.17E-04	8.39
Ozone formation, Terrestrial ecosystems	9.10E-03	7.65E-04	8.41
Stratospheric ozone depletion	1.10E-02	2.62E-03	23.76
Terrestrial acidification	5.63E-03	5.17E-04	9.17
Terrestrial ecotoxicity	2.15E-01	9.16E-02	42.60
Water consumption	4.67E-03	4.32E-02	924.50

A.7. Sensitivity analysis

The bio-composite default composition as communicated by manufacturer is 58% calcite, 27% Polyester resin and 15% water reeds by mass. At this early stage of product development, a variety of different compositions may be attempted and lead to viable products. To evaluate how different compositions can affect the environmental performance, a sensitivity analysis is conducted. The three component fractions are increased by 10% one at a time. The other two fractions are reduced by 5% each. Figure A.3 shows only a few selected categories, for ease of display. Fibre content initially is assumed around 15%, but, it can be as high as 30-40% (Ita-Nagy et al., 2020). It was thus, increased to 25% and the other two reduced by 5% each. Upon increasing the reed content by 10%, the largest percentage improvement is seen for the category of stratospheric ozone depletion (40% reduction). Ozone depletion is found to be mainly caused by dinitrogen monoxide emissions from polyester resin manufacturing. Similarly, calcite fraction is increased from 58% to 68% and the largest improvement is again seen for stratospheric ozone depletion (40% reduction).

Generally speaking, the impact reduction from increasing reed content or calcite content by 10% each is very similar for all categories, but the largest reduction of around 40% is seen for stratospheric ozone depletion. Overall, increasing either of the components can be a way to reduce environmental negative impacts as long as the resin content is reduced.

A.8. Interpretation

The most significant issue is the polyester resin manufacturing. It accounts for over 50% of all impact categories except for freshwater eutrophication

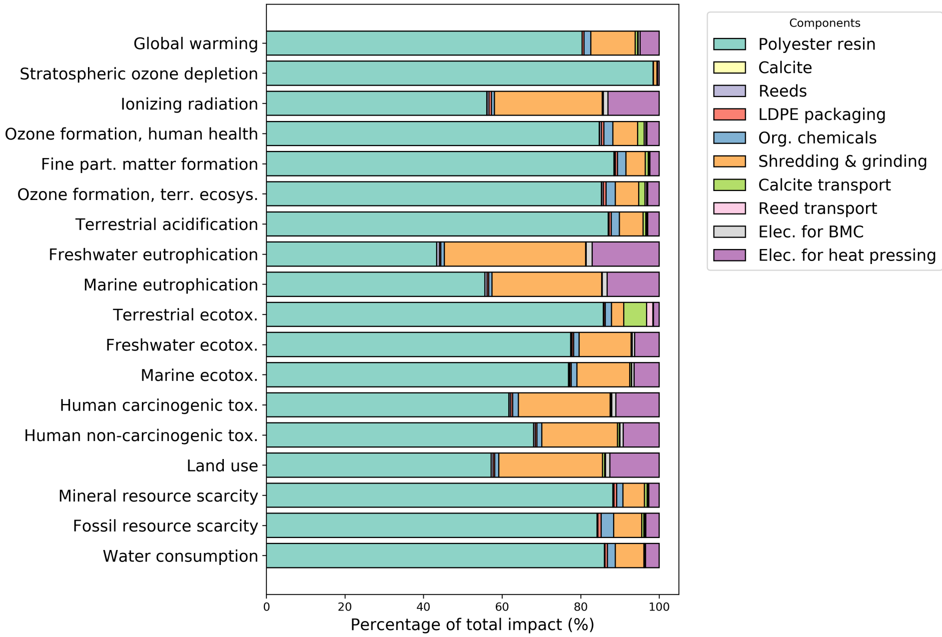

Figure A.2: Contribution (%) of various life cycle stages to the overall environmental damage. Polyester resin production and recycling of the bio-composite are found to be the largest contributors. A small negative emission for global warming results from the reeds storing CO_2.

(FE). For FE, the contribution of polyester resin is 43% and that of shredding and grinding for recycling stands at 36%. The impacts are highly sensitive to the quantity of resin used. Lowering its proportion and increasing calcite and/or reeds is recommended for reducing the negative impacts of the composite. The second most significant impact is due to the recycling process of the bio-composite. The high environmental negative impact of the recycling process is mainly due to high electricity consumption. Mechanical grinding is assumed as the process used for recycling the bio-composite. An average energy consumption value of 2.45 MJ/kg is assumed based on Buggy et al. (1995). As source of this energy, electricity fuel mix of Germany for the year 2016 is assumed which is around 40% coal-based. Emissions from lignite mining are seen the major culprit to the negative impacts of electricity production. Making use of non-fossil-based electricity source can be expected to further improve its environmental performance significantly.

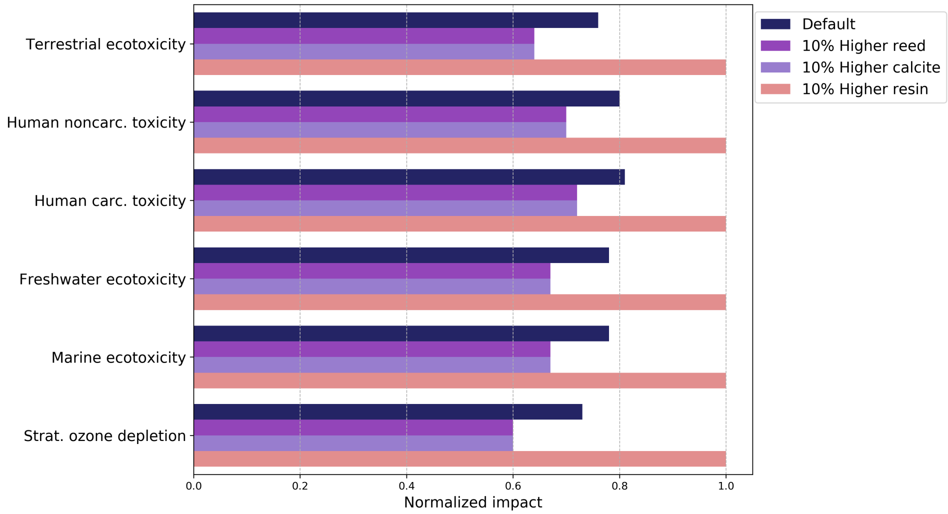

Figure A.3: Sensitivity analysis results for four compositions; default: 58% Calcite, 27% Resin, & 15% Reed; 10% Higher reed: 53% Calcite, 22% Resin, & 20% Reed; 10% Higher calcite: 68% Calcite, 22% Resin, & 10% Reed; 10% Higher resin: 53% Calcite, 37% Resin, & 10% Reed. The results are normalized to the maximum impact composition (10% Higher resin).

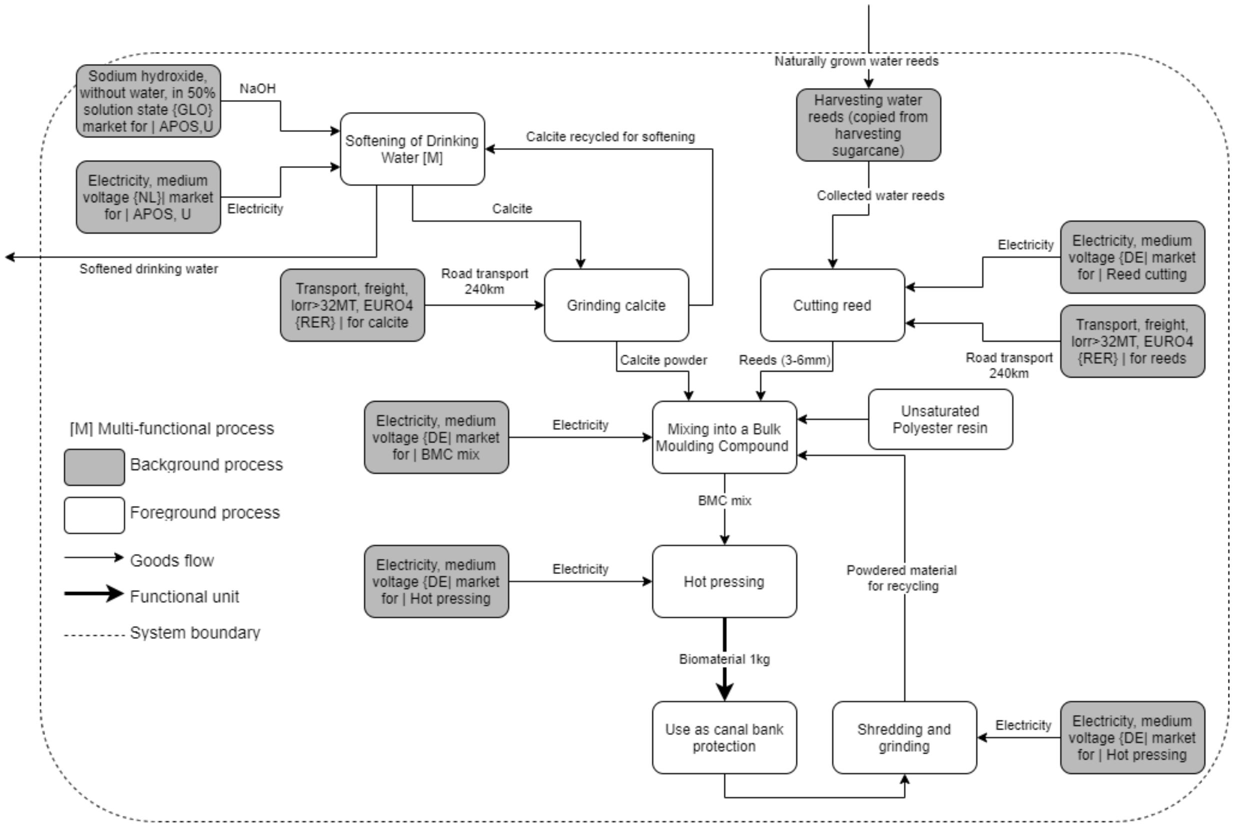

Figure 6.4: A flow-chart for the cradle-to-cradle LCA of the bio-composite's application as canal bank protection. The system boundary begins with raw material extraction from the urban water system. The bio-composite is used as canal bank protection. The last process is shredding and grinding the material to use as feedstock for new production.

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