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Pathways to future sustainable wastewater management

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ABSTRACT

Wastewater management (WWM) plays a crucial role in protecting the environment and public health while also contributing to sustainable development. WWM currently faces significant challenges driven by ageing infrastructure and urbanisation, as well as uncertainties about future developments related to regulations, climate impact, energy use, and resource recovery. Developing WWM sustainably therefore requires decision support that can account for such challenges and uncertainties while balancing environmental, social, and economic considerations. This thesis addresses three issues: which future conditions WWM should prepare for, what sustainable WWM entails, and how alternative development pathways can be assessed in terms of their sustainability.

First, to identify future conditions that WWM should prepare for, a future scenario approach was developed, combining stakeholder participation, expert knowledge, and process modelling. Applied to an existing wastewater treatment plant (WWTP), the approach identified and specified relevant future conditions related to flows, treatment requirements, and environmental and climate impact. Process modelling of quantified future scenarios showed that the WWTP would require substantial upgrades to meet anticipated future demands. Second, to clarify what sustainable WWM entails, a definition of sustainable WWM was formulated by interpreting and synthesising established sustainability frameworks in the context of WWM. The resulting definition consists of eleven sustainability areas, specified by objectives and linked to criteria, providing a structured basis for understanding and assessing sustainable WWM. Third, to compare alternative development pathways, a sustainability assessment method based on multicriteria analysis was developed. The methodological contributions include linear scoring, calculation of weighted score ratios, structured sensitivity analyses, and a monetary-based weighting procedure supported by a compilation of monetary reference values. These contributions aim to increase transparency and help decision-makers understand how scoring and weighting choices affect results. Application of the method to case studies showed that making monetary implications explicit and relating them to reference values facilitates discussion of criterion weighting and trade-offs. Including sensitivity analyses helps evaluate result robustness and account for uncertainties.

This thesis provides a coherent framework for sustainable decision-making in WWM, with interconnected components that can be used individually. The thesis supports decision-making under uncertainty and promotes transparent and robust planning of future WWM.

Keywords: wastewater treatment plant; wastewater systems; decision support; future scenario analysis; sustainability definition; sustainability assessment; multicriteria analysis; weighted score ratios; monetary valuation.

LIST OF PUBLICATIONS

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

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Paper I: Investigation; Formal analysis; Data curation; Visualization; Writing - Original draft; Writing - Review & editing.

Paper II: Conceptualisation; Methodology; Investigation; Visualization; Writing - Original draft; Writing - Review & editing.

Paper III: Conceptualisation; Methodology; Investigation; Formal analysis; Data curation; Visualization; Writing - Original draft; Writing - Review & editing.

Paper IV: Conceptualisation; Methodology; Investigation; Formal analysis; Data curation; Visualization; Writing - Original draft; Writing - Review & editing.

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This project began with a broad scope, but along the way, we have worked together to develop something practically useful that will hopefully lead to more sustainable wastewater management in the future. Thank you all for contributing to this.

ABBREVIATIONS

AGS	Aerobic Granular Sludge
AHP	Analytic Hierarchy Process
AS	Activated Sludge
ASM1	Activated Sludge Model No. 1
BOD_5	Biochemical oxygen demand over 5 days
BOD_7	Biochemical oxygen demand over 7 days
BSAP	Baltic Sea action plan
CBA	Cost benefit analysis
CH_4	Methane
CO_2e	Carbon dioxide equivalent
CSO	Combined sewer overflow
CV	Coefficient of variation
EIA	Environmental impact assessment
EU	European Union
EUR	Euro, €
GAC	Granular Activated Carbon
GHG	Greenhouse gas
IFAS	Integrated Film Activated Sludge
LCA	Life cycle analysis
LCC	Life cycle costing
LCOE	Levelized cost of energy
MAUT	Multi-Attribute Utility Theory
MBR	Membrane Bio Reactor
MBBR	Moving Bed Biofilm Reactor
MCA	Multicriteria analysis
MCDA	Multicriteria Decision Analysis
MDG	Millennium Development Goal
N	Nitrogen
N_2O	Nitrous oxide
P	Phosphorus
PAC	Powdered Activated Carbon
PAOs	Polyphosphate-accumulating organisms
pe	Population equivalent
Δs	Range in scores
SDG	Sustainable Development Goal
SEK	Swedish kronor
TF	Trickling Filter
UN	United Nations
USD	US dollars
Δv	Range in values
V_i	Value per score step for criterion i
w_i	weight for criterion i
WCED	World Commission on Environment and Development
WSR	Weighted score ratios
WWM	Wastewater management
WWT	Wastewater treatment
WWTP	Wastewater treatment plant

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1 Introduction

This chapter presents the research aim and objectives, outlines the scope and delimitations of the study, and provides an overview of the thesis structure.

1.1 Research context

Wastewater systems represent major investments and fundamental infrastructure in modern societies, designed to operate for many decades (Lienert et al., 2006). Across Europe, a large share of this infrastructure was constructed decades ago and is now ageing (Braga et al., 2026b), creating a growing need for renovation and replacement. Given their long lifespan and high economic value, decisions regarding renovation, upgrades, and new construction require well-informed planning to ensure effective and relevant solutions. This increasing investment demand in the wastewater sector is driven not only by ageing infrastructure but also by new and stricter regulatory requirements (EU, 2024). These regulations address both conventional pollutants, including organic matter, nitrogen (N), and phosphorus (P), and micropollutants, such as pharmaceutical residues. At the same time, urban populations continue to grow, and many wastewater systems are approaching or exceeding their design capacity (Braga et al., 2026a). There is also growing awareness of the wider environmental impacts of wastewater management (WWM), including climate impact and energy use (EU, 2024, Rockström et al., 2009, UN, 2026).

Collectively, these developments are placing increased pressure on WWM and have brought sustainability high on the wastewater planning agenda. Decision-makers are expected to ensure that investments not only address immediate challenges but also contribute to sustainability goals at multiple levels, from the Sustainable Development Goals (SDGs) (UN, 2026) to local objectives, all within economic constraints. These factors are driving a wave of large-scale WWM projects (Tchobanoglous et al., 2014). Planning these projects, and thus future WWM, requires consideration of a wide range of often uncertain factors. Understanding how regulatory requirements, wastewater flows, and pollutant loads may develop is therefore critical for long-term planning. They are in turn influenced by changes in legislation, population growth, lifestyle patterns, and stormwater management. Climate change adds complexity through changing precipitation patterns and the need for climate adaptation (Butler et al., 2017). In addition, the cost, availability, and sustainability of energy and chemicals are key considerations in planning.

Many of these challenges and developments in the wastewater sector are highly relevant to Gryaab, the municipal company operating the Rya wastewater treatment plant (WWTP) in Gothenburg, Sweden. Rya WWTP is one of the largest plants in northern Europe, making its future development particularly important. In this context, and as part of Gryaab's ambition and continued engagement in research and development, the industrial PhD project *Pathways to future sustainable wastewater management* was initiated. The project aimed both to advance research in the field and to support the long-term strategic planning of Gryaab and other WWTPs, where these issues are central. Several developments that were uncertain at the start of the project have since been formalised through the revised Urban Wastewater Treatment Directive, which emphasises increased nitrogen removal, micropollutant removal, energy neutrality, and reduced climate impact (EU, 2024). This regulatory shift further underscores the relevance of the research presented in this thesis.

1.2 Aim and objectives

The overarching aim of this thesis is to identify and explore pathways to future sustainable wastewater management. Because this is a broad aim that can be approached in different ways, at both process and system level, it requires further specification. A central task is to clarify what constitutes a sustainable pathway within WWM and how such pathways can be identified, with pathways referring to alternative choices or directions for development. Planning for the future requires consideration of uncertain future conditions, while sustainability implies that the pathways must be systematically evaluated.

To address this, the work is structured around three key components: (i) the future conditions in which WWM will operate, (ii) the meaning of sustainability within WWM, and (iii) how the sustainability of pathways can be identified and assessed. Based on these key components and to specify the overarching aim, the thesis has the following research objectives:

1. **Develop an approach for identifying future conditions for WWM** through scenario analysis, capturing a wide range of aspects and sustainability priorities to support well-informed planning of future WWM.
2. **Provide a basis for sustainability assessments by defining sustainable WWM** through a clear and detailed specification grounded in scientific literature and well-established sustainability frameworks.
3. **Develop a practical sustainability assessment method for WWM** that facilitates transparent and consistent comparison of alternatives, incorporates relevant sustainability aspects and future uncertainties, and enables evaluation of trade-offs and the robustness of the results.

1.3 Scope of the work

The thesis is structured into three steps, addressed in the four included papers.

Step 1: Future conditions for WWM (Paper I)

The first step establishes the future context for WWM by developing and applying a scenario-based approach. Future conditions are identified and incorporated into scenarios, including changes in regulations, population development, lifestyle patterns, climate, and wastewater systems. These scenarios provide the conditions under which sustainable pathways must operate.

Step 2: Definition of sustainable WWM (Paper II)

The second step defines sustainable WWM by examining how sustainability is conceptualized in the literature and how key sustainability frameworks can be interpreted for WWM. The resulting definition provides a foundation for assessing and comparing alternative pathways for future WWM in terms of their sustainability.

Step 3: Development of a sustainability assessment method for WWM (Paper III and Paper IV)

The third step develops a practical sustainability assessment method for WWM. Building on an existing methodological approach and the sustainability definition from Step 2, the method provides systematic decision support for comparing alternative pathways. The method is applied, evaluated, and refined through WWM case studies, with particular emphasis on improving its practical applicability, transparency and consistency.

1.4 Delimitations

- The system boundaries are primarily limited to the wastewater system, with a focus on the WWTP rather than the entire urban water system.
- Future scenarios focus on conditions most relevant to WWM rather than all possible developments. Future scenarios are developed using a single methodological approach, and alternative scenario development methods are not thoroughly explored.
- The review of sustainability assessment methods is not comprehensive but provides an overview and supports the selection of a method for further development.
- The case studies are used to develop, demonstrate, evaluate, and refine the proposed assessment method. The alternatives assessed are not developed within this thesis, and their real-world implementation is outside the scope of the thesis.
- These case studies are conducted in a Swedish context, although the method is intended to be applicable across different contexts.
- The sustainability assessment method's applicability beyond WWM is not evaluated.

1.5 Thesis structure

The thesis structure follows the main steps of the research process and is organized as follows:

Chapter 1 – Introduction

Presents the research context and outlines the aim, objectives, scope, and delimitations.

Chapter 2 – Background

Provides background on WWM, sustainability, and sustainability assessment methods.

Chapter 3 – Future scenarios for WWM

Presents a scenario-based approach for identifying future conditions relevant to WWM and discusses their implications for future planning.

Chapter 4 – Definition of sustainable WWM

Defines sustainable WWM based on scientific literature and well-established frameworks, providing the foundation for sustainability assessment.

Chapter 5 – Sustainability assessment method for WWM

Presents the development and applications of a sustainability assessment method for evaluating alternative WWM pathways.

Chapter 6 – Overall discussion

Discusses the combined findings from Chapters 3-5 and their implications for decision-making and pathways to future sustainable WWM.

Chapter 7 – Conclusions and future work

Summarizes the main conclusions of the thesis and identifies possible directions for future work.

Paper I–IV

The papers are included in full at the end of the thesis.

2 Background

This chapter presents the background to the thesis, providing a basis for understanding its content. It begins with an overview of WWM, followed by a section on the concept and history of sustainability and sustainable development, and concludes with an introduction to sustainability assessment methods.

2.1 Wastewater management

This section provides a general overview of wastewater systems in developed countries, serving as a foundation for identifying pathways toward more sustainable systems.

In this thesis, the term WWM is generally used to highlight a broader perspective than WWTPs alone. However, while WWM can include the entire chain from wastewater generation to the receiving environment for the treated effluent, the primary focus of this work is still on activities and impacts associated with the WWTP. The current structure of wastewater systems has developed gradually over time and serves as the starting point for this thesis, while recognising that this structure may need to be questioned and changed. Wastewater systems vary considerably worldwide, and this thesis primarily reflects conditions at larger WWTPs in developed countries, although some studies are more broadly applicable.

2.1.1 History of wastewater management

The historical development of WWM helps to explain the structure of current systems. The evolution of WWM can generally be described as a series of stages reflecting changing societal priorities and an increasing understanding of the environmental and other impacts that WWM causes. Early developments of wastewater systems are still largely embedded in current systems, highlighting the importance of making well-informed decisions for future development, which forms the core of this thesis.

Sanitation and public health protection

Early developments in WWM can be traced back to ancient civilizations such as the Greeks and the Roman Empire (Lofrano and Brown, 2010). The first steps of WWM focused on sanitation and the protection of public health (Tchobanoglous et al., 2014). Wastewater and human excreta had long been regarded as both a health risk and a source of unpleasant odours in urban areas. As wastewater was collected and transported away from densely populated areas, human exposure to pathogens was reduced and urban hygiene improved. In a modern context, when sewer systems for collecting the wastewater began to appear in many of Europe's larger cities, wastewater was typically discharged untreated directly into nearby waters or onto land (Chen et al., 2020).

The introduction of wastewater treatment

With industrialization came rapid urban growth, making the environmental impacts of untreated wastewater increasingly apparent. Receiving waters could not cope with the growing wastewater loads and began to show clear signs of pollution. The quality of water bodies near urban areas worsened and became problematic, leading to bathing restrictions in some locations, widespread oxygen depletion affecting aquatic life, and reduced availability of clean water resources (Daigger, 2009, Svenskt Vatten, 2010b). In response, wastewater treatment (WWT) was introduced, initially with physical methods inspired by natural processes, such as screening and sedimentation (Tchobanoglous et al., 2014). On a larger scale, WWT was implemented in the early twentieth

century in large European cities but its development was disrupted by the two World Wars (Lofrano and Brown, 2010). As regulatory requirements became stricter, such as with the Clean Water Act of 1972 in the US, additional pollutants were regulated and new treatment processes were developed and implemented (Chen et al., 2020, Lofrano and Brown, 2010).

Much of the development of WWT, like other forms of environmental protection, has been driven by an anthropocentric perspective, primarily motivated by the importance of the environment for human health and well-being rather than for its intrinsic value (Murray, 2017).

Eutrophication mitigation and centralization of wastewater treatment

During the second half of the twentieth century, eutrophication emerged as a major environmental concern in surface waters. Eutrophication refers to nutrient enrichment, primarily by nitrogen and phosphorus, which can lead to excessive growth of algae and aquatic plants. It became clear that reductions in nutrient discharges and stricter regulations were needed to limit eutrophication. This led to a general upgrading of WWTPs (Chen et al., 2020, Svenskt Vatten, 2010b). The upgrading was partly achieved through further development of biological treatment, originally introduced to improve the removal of organic matter and later to remove phosphorus and nitrogen (Tchobanoglous et al., 2014). One of the most important developments was the activated sludge process in the United Kingdom (Ardern and Lockett, 1914). The process was developed in the early twentieth century and was subsequently refined and became widespread as infrastructure expanded and environmental regulations strengthened. It eventually became the standard biological treatment method at large WWTPs. Today, it remains a crucial biological treatment method worldwide (Jenkins and Wanner, 2014).

To effectively mitigate eutrophication, additional phosphorus removal was also needed in some locations and as a result, chemical treatment methods were introduced, with phosphorus removed by the addition of precipitation chemicals, followed by sedimentation or filtration (Chen et al., 2020). Increasing treatment requirements also contributed to the centralization of WWT in urban areas. Smaller treatment plants were converted into pumping stations to transfer wastewater to larger, centralized WWTPs (Svenskt Vatten, 2010b). This centralization has been largely driven by economies of scale (Pasciucco et al., 2022). In Europe, the adoption of the Urban Wastewater Treatment Directive (EU, 1991) strongly influenced the development of WWT. The directive established minimum requirements for urban wastewater collection and treatment across European Union (EU) member states, leading to substantial infrastructure investments.

Broader focus on indirect consequences and micropollutants

The treatment requirements have continued to become successively stricter, alongside a growing focus on the indirect consequences of WWM. Pollution control remains central, but energy use, climate impact, and resource use have become increasingly important (Daigger, 2009). Attention is also directed towards micropollutants, such as pharmaceutical residues (Burzio et al., 2022). These aspects are further discussed in Section 2.1.4.

This description represents a general historical development pattern for developed countries. In practice, WWM has evolved gradually and unevenly, with variation between countries, regions, and WWTPs depending on local conditions, regulatory requirements, and other factors. While the historical development of WWM provides context for its current state, it also demonstrates the dynamic nature of WWM (Dominguez and Gujer, 2006), highlighting that it is likely to continue evolving in response to new challenges.

2.1.2 Wastewater systems: An overview

Wastewater systems can be categorized as centralized or decentralized with current wastewater systems being predominantly centralized (Söderholm et al., 2023). In Sweden, many small plants serve rural communities, while fewer large plants serve major urban areas (Svenskt Vatten, 2026). A similar pattern is observed across Europe, where plant sizes range from very small facilities to large regional plants for several hundred thousand people (Ganora et al., 2019). While this thesis primarily focuses on larger WWTPs, many of the principles discussed are also applicable to smaller plants.

In centralized systems, wastewater is collected and transported through sewer networks to WWTPs. Sewer systems may be either combined or separate, where combined systems collect both wastewater and stormwater in the same network, whereas separate systems convey them in different pipes (Sperling, 2007). To improve treatment efficiency and reduce environmental impacts, separate systems are generally preferred in modern infrastructure, as they reduce hydraulic loading at treatment plants, limit the risk of untreated discharges during heavy rainfall, and avoid contamination of wastewater with stormwater (Svenskt Vatten, 2010b).

Influent to a WWTP typically comprises domestic wastewater, wastewater from commercial activities and institutions (e.g. hospitals and schools), industrial wastewater, infiltration into sewer systems, stormwater, landfill leachate, septic tank wastewater, and internal WWTP streams (Tchobanoglous et al., 2014, Chen et al., 2020). Combined sewer systems receive a larger proportion of stormwater than separate systems. Wastewater characteristics are further influenced by sewer hydraulic retention time, as well as by seasonal, weekly, daily, and local conditions (Sperling, 2007). Influent wastewater contains organic matter, nutrients (such as nitrogen and phosphorus), microorganisms, metals, inorganic substances, and micropollutants, but also odour and heat (Jansen et al., 2019). Industrial wastewater may require pretreatment before being discharged to the sewer network (Tchobanoglous et al., 2014). Overall, wastewater contains both valuable resources, such as energy and nutrients, and potentially harmful substances that need to be carefully managed.

Although WWTPs differ in design and complexity, a general treatment line can typically be identified for larger WWTPs, reflecting the gradual development of WWT described in Section 2.1.1. WWT can typically be divided into preliminary, primary, secondary, tertiary, quaternary, and sludge treatment (Tchobanoglous et al., 2014). In European countries, discharge limits have focused on the removal of organic matter, nitrogen, and phosphorus to protect receiving waters from oxygen depletion and eutrophication (EU, 1991). More recently, increasing attention has been directed toward the removal of micropollutants, as discussed in 2.1.4.

In the **preliminary treatment,** coarse materials and particles are separated from the wastewater, primarily to protect the equipment in the subsequent treatment steps (Svenskt Vatten, 2010a). This stage typically includes coarse and fine screening, grit removal, and oil and grease separation (Englande et al., 2015). These initial treatment steps require energy for operating mechanical equipment, and for pumping of the influent wastewater (Olsson, 2011). The separated materials are usually dewatered and incinerated (Svenskt Vatten, 2010a).

Primary treatment reduces the load of suspended solids in the wastewater, typically through sedimentation in primary clarifiers, with microscreening, flotation, and filtration as alternative methods (Tchobanoglous et al., 2014). In conventional sedimentation tanks, fats and lighter materials float to the surface while heavier solids settle at the bottom. Both fractions are mechanically removed, and the resulting primary sludge is sent to sludge treatment.

Primary sedimentation tanks can also be used for stormwater retention and partial treatment with chemical precipitation, during high-flow events (Tchobanoglous et al., 2014). Suspended solids, which contain particulate organic matter and some nitrogen and phosphorus, are removed with the sludge, whereas dissolved fractions, containing most of the nitrogen and phosphorus, generally remain in the wastewater (Svenskt Vatten, 2010b). The removal efficiency of suspended solids and phosphorus can be enhanced through the addition of precipitation chemicals, typically a metal salt, and possibly polymer (Englande et al., 2015). Thus, primary treatment may require both energy and chemicals, with energy used to operate scrapers and pumps and chemicals used to enhance solids and phosphorus removal. At the same time, the organic carbon captured in the sludge represents a potential energy resource that can be recovered, for example through anaerobic digestion (Capodaglio and Olsson, 2020).

Following primary treatment, **secondary and tertiary treatment** aim to remove organic matter and often nitrogen, phosphorus, and pathogens. These stages typically involve biological processes involving different microorganisms. Biological treatment can broadly be classified into suspended growth or attached growth (fixed-film) processes, or a combination of both, in which microorganisms either grow in suspension or on surfaces (Tchobanoglous et al., 2014). Examples include activated sludge (AS), moving bed biofilm reactors (MBBR), trickling filters (TF), integrated fixed-film activated sludge (IFAS), aerobic granular sludge (AGS), and membrane bioreactors (MBR). The activated sludge process requires subsequent solids separation, typically achieved through sedimentation in secondary clarifiers (Jenkins and Wanner, 2014). Solids separation is also often required in attached growth systems to remove excess biomass. In activated sludge systems, part of the settled sludge is recirculated to the biological process, while the excess sludge (waste activated sludge) is sent to sludge treatment (Tchobanoglous et al., 2014).

Organic matter is primarily removed through oxidation to carbon dioxide and assimilation into biomass. For this oxidation, either oxygen (under aerobic conditions), or nitrate (under anoxic conditions) can be used. Nitrogen in the influent wastewater is present mainly as ammonia and organically bound nitrogen. Ammonia is oxidized to nitrate under aerobic conditions through nitrification. This is followed by denitrification, in which nitrate is reduced to nitrogen gas that is released to the atmosphere. During the denitrification, organic matter is also oxidized (Rahimi et al., 2020).

Phosphorus removal can be achieved by designing the activated sludge process for biological phosphorus removal, in which microorganisms are exposed to alternating anaerobic and aerobic conditions. This promotes the activity of polyphosphate-accumulating organisms (PAOs), which take up and store phosphorus in the biomass, leading to its removal with the sludge. An alternative or complementary method for phosphorus removal is chemical precipitation, achieved through the addition of metal salts. This can be applied either within the biological process (simultaneous precipitation) or before (pre-precipitation) or after biological treatment (post-precipitation), followed by separation of the resulting sludge (Tchobanoglous et al., 2014).

The aeration of biological treatment processes is a significant energy input (Olsson, 2011), while some energy is also required for mixing in the tanks. When strict nitrogen discharge limits apply, an external carbon source may be added to enhance nitrogen removal in the denitrification step (Chen et al., 2020). Potential climate impact from the biological treatment includes the use of fossil-based external carbon sources (Åmand et al., 2015, Gustavsson and Tumlin, 2013) and emissions of nitrous oxide from the biological reactions (Massara et al., 2017). Further treatment may be applied to remove remaining suspended solids, dissolved organic and inorganic matter, and pathogens from the effluent in order to meet discharge or reuse standards.

Common processes include filtration methods, such as sand or disc filters, as well as disinfection methods, including ultraviolet radiation, ozonation, or chlorination. Energy and chemical requirements depend on the technologies applied (Tchobanoglous et al., 2014).

Quaternary treatment, sometimes referred to as advanced treatment, constitutes an additional treatment stage aimed at reducing micropollutants, such as pharmaceutical residues, personal care products, pesticides, synthetic and natural hormones, that are not effectively removed by conventional treatment methods (Bui et al., 2016). Typical methods include ozonation and activated carbon (Bui et al., 2016, Margot et al., 2013). This stage is increasingly being implemented, partly driven by emerging regulatory requirements, for example within the EU (EU, 2024). However, these processes are often energy-intensive and may have a high climate impact, particularly when fossil-based activated carbon is used (Pistocchi et al., 2022).

Sludge treatment is needed to reduce sludge volume and water content, stabilize organic matter, enable energy recovery and nutrient recycling, and ensure safe handling, disposal, or reuse (Svenskt Vatten, 2010b). Sludge typically originates from primary clarifiers, biological treatment, and chemical precipitation with characteristics depending on the wastewater composition, treatment processes and sludge retention time (Tchobanoglous et al., 2014). Sludge treatment typically starts with thickening, to reduce the water content and concentrate the solids. This is often followed by stabilization, which for larger plants typically means anaerobic digestion to reduce odour and pathogens and convert organic matter into biogas, which can be utilized as an energy source. Subsequently, dewatering further reduces the water content, after which the dewatered sludge can be hygienised by different methods. Chemical additives, such as polymers, are often used to enhance dewatering, while energy is required to operate mechanical equipment and heat the digestor. In some cases, additional treatment steps, such as drying, thermal treatment, or incineration, are applied (Tchobanoglous et al., 2014).

Sludge treatment plays an important role in energy and resource recovery. It enables the recovery of energy from organic matter, primarily through biogas production, and nutrients such as phosphorus and nitrogen. At the same time, sludge treatment must address potential risks related to contaminants and pathogens to ensure safe and sustainable reuse or disposal (Capodaglio and Callegari, 2023).

Internal wastewater streams from sludge treatment include thickener supernatant and reject water from dewatering (Jansen et al., 2019). These streams can be returned to the main treatment line, but reject water from dewatering is more commonly treated separately in a sidestream process (Stenström et al., 2017). Its high ammonia concentration and relatively high temperature provide favourable conditions for energy-efficient nitrogen removal through partial nitritation-anammox. Compared with conventional nitrification-denitrification, this process requires less aeration energy and no external carbon source (Stenström et al., 2017).

2.1.3 Rya wastewater treatment plant

As Rya WWTP is included in several case studies, a brief description of the plant and its operational context is provided. Rya WWTP is located in Gothenburg, Sweden, and is operated by Gryaab AB, a company jointly owned by eight municipalities in the region: Ale, Gothenburg, Härryda, Kungälv, Lerum, Mölndal, Partille, and Bollebygd. In 2025, the plant received wastewater corresponding to approximately 840 000 population equivalents (pe), transported to the plant through a tunnel system approximately 130 km long. Part of the sewer system is combined, and infiltration and inflow contribute to large variations in influent flow, ranging from approximately 2 to 16 m^3/s, with an average flow of 3.8 m^3/s in 2025 (Videbris, 2026).

An overview of the treatment process at Rya WWTP is shown in Figure 1. It is fully automated and can be broadly divided into mechanical, chemical, and biological treatment. Mechanical treatment includes coarse and fine screening, grit removal, and primary sedimentation. Chemical treatment involves the addition of iron sulphate and polymer to form flocs that are subsequently removed by sedimentation. Biological treatment includes high-loaded activated sludge for pre-denitrification, organic matter removal, and simultaneous phosphorus precipitation. This is followed by nitrification in either trickling filters or MBBRs, and post-denitrification in MBBRs with the addition of an external carbon source. Final polishing is achieved through disc filtration. Heat is recovered from the treated effluent using heat pumps operated by a municipal energy company and utilised in the district heating system. The treated effluent is discharged into the Göta Älv estuary, where dilution is relatively high. The current discharge limits for Rya WWTP are specified in Table 1 and include 7-day biochemical oxygen demand (BOD_7), total phosphorus, and total nitrogen.

Table 1. Current discharge limits for Rya WWTP for BOD_7, total phosphorus, and total nitrogen. Concentration limits are based on annual and May–August averages, and load limits on rolling three-year averages.

Parameter	**Concentration limit (mg/L)** *annual average and May-August average*	**Load limit (tonnes/year),** *(rolling 3-year average)*
BOD_7	10	1300
Total phosphorus	0.3	40
Total nitrogen	8	1000

The sludge treatment line includes thickening, mesophilic anaerobic digestion for biogas production, and dewatering. The resulting sludge is recycled to agricultural land after hygienisation, provided that it meets the requirements of the Revaq certification system, as described by Mattsson et al. (2017). In 2025, 100% of the sludge was recycled (Videbris, 2026) compared with an average of 84% over the past five years. Reject water from dewatering is primarily treated using an MBBR anammox process and is then directed to the nitrifying MBBR, after which it follows the main treatment line.

Rya WWTP was commissioned in 1972 and has since undergone continuous upgrades and expansions. Due to limited space, Gryaab has developed a compact treatment plant capable of handling high and variable inflows while meeting regulatory requirements. Plant performance is optimised and flow variability managed through flexible process design and operational strategies, including the use of primary clarifiers for stormwater treatment, dynamic control of the tunnel system for flow equalisation, dynamic process modelling, and continuous optimisation of sedimentation processes to maximise capacity (Wilén et al., 2010, Hanner et al., 2004).

Looking ahead, Gryaab is carrying out a long-term major expansion project, aimed at meeting future demands, stricter regulations, and a growing population. The expansion, initiated in 2019 and expected to be completed by 2036, will take place on adjacent land. Gryaab is also in the process of obtaining a new environmental permit, which is expected to introduce stricter regulatory requirements and discharge limits. In addition, improving resource and energy efficiency, reducing climate impact, and maintaining cost-effectiveness are key priorities, with the overall aim of developing a more sustainable WWTP.

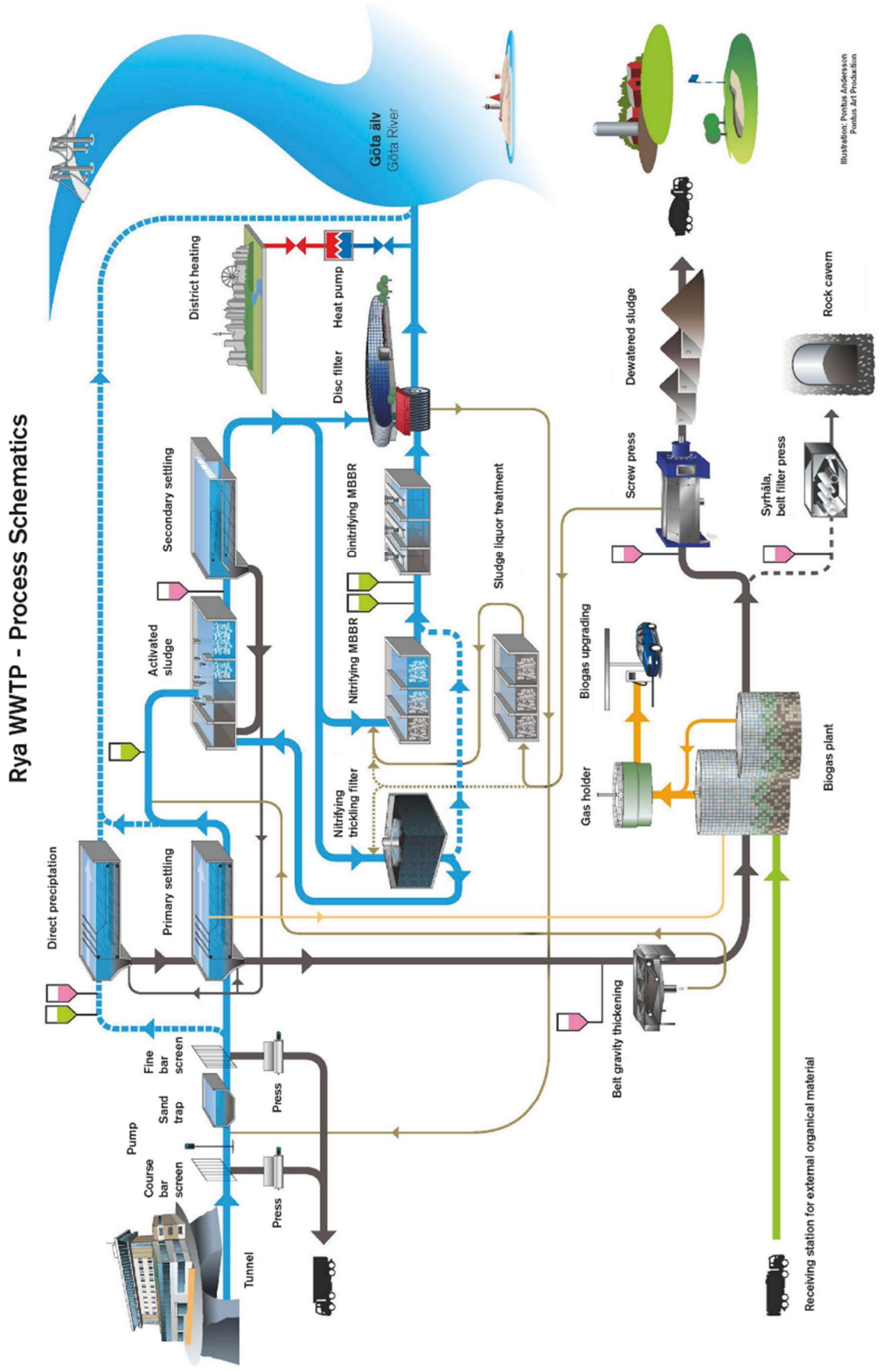


Figure 1. Process schematic of the Rya WWTP in Gothenburg, Sweden.

2.1.4 Opportunities and challenges

WWM has improved significantly in recent decades in developed countries, where centralized systems are the conventional approach (Söderholm et al., 2023) and highly developed treatment technologies are widely implemented. Nevertheless, major challenges remain. Globally, around 3 billion people are estimated to still lack access to safely managed sanitation by 2030, and a substantial share of the world's wastewater is not adequately treated, posing risks to human health and the environment (UN, 2024). Environmental degradation of recipients has led to increasingly stringent regulations, targeting both conventional pollutants and micropollutants, as outlined in Section 1.1. In addition, the widespread occurrence of antibiotics in wastewater and receiving waters is an environmental and public health concern, highlighting the need for further action, including source reduction and measures to address bypass flows such as combined sewer overflows (CSOs) (Bekele et al., 2026, Lee et al., 2022).

Resource recovery, particularly of energy and nutrients, represents a key opportunity in modern WWM that is largely underutilized (Lehtoranta et al., 2022, UN, 2017). At the same time, challenges remain, for instance in sludge management, where nutrient recycling needs to be balanced with potential risks of spreading contaminants (Pozzebon and Seifert, 2023).

Other challenges identified in Section 1.1 include ageing infrastructure, population growth, climate change, and treatment systems operating close to their capacity limits, alongside increasing sustainability ambitions. The need to upgrade and expand existing systems creates opportunities to redesign WWM towards more sustainable and efficient solutions, including a focus on energy efficiency and reduced climate impact of WWTPs (Sarpong and Gude, 2020, Larsen, 2015). With advances in knowledge, technologies, and approaches, there is potential to develop robust, adaptable systems capable of meeting future demands and stricter regulations.

2.2 The concept of sustainable development

Given the many challenges facing WWM, the transition towards sustainable systems is increasingly important. While WWM constitutes the core area of this thesis, an understanding of the conceptual foundations and historical evolution of sustainability and sustainable development is equally essential. This section therefore provides an overview of sustainable development and important sustainability frameworks, before relating it more specifically to WWM in later chapters.

2.2.1 History of sustainable development

Hedenus et al. state that "to understand the concept of sustainable development, we first have to understand its historical roots" (Hedenus et al., 2022). A key element of these roots is the idea of continuous progress, understood as a movement in a desirable direction for civilization (Du Pisani, 2006). This idea of progress had already emerged in the Greco-Roman period but became more central during industrialization, when economic growth and material advancement were seen as pathways to improved human well-being (Du Pisani, 2006). Over time, it became clear that such progress was difficult to maintain, as it was accompanied by growing environmental degradation and secularization (Du Pisani, 2006, Purvis et al., 2019).

Humans have historically influenced their local environments through activities such as hunting, farming, logging, and mining (Hedenus et al., 2022, Du Pisani, 2006). However, it was after World War II, during the "Great Acceleration", that humans' impact on the environment intensified and became more widespread (Hibbard et al., 2006). Environmental concerns gained increased attention during the 1960s and 1970s (Purvis et al., 2019) and Hedenus et al. (2022) note that this

growing awareness was strongly influenced by the book *Silent Spring* (Carson, 1962), followed by the report *The Limits to Growth* (Meadows, 1972). *Silent Spring* addressed the environmental damage caused by chemical pesticides and *The Limits to Growth*, which was based on simulations from the computer model *World3*, warned that unchecked economic and population growth could exceed the planet's limits and lead to collapse.

At the same time, a social critique emerged that pointed out that economic growth often failed to address poverty, inequality, and basic human needs (Purvis et al., 2019). Together, the growing recognition of environmental degradation and the *Limits to Growth* questioned the traditional view of development (Purvis et al., 2019, Du Pisani, 2006). These new perspectives sought to integrate environmental protection, social equity, and economic considerations, and in this context, organisations such as Friends of the Earth and Greenpeace were established (Hedenus et al., 2022).

In 1972, the United Nations (UN) Conference on the Human Environment was held in Stockholm, the first conference of its kind where environmental protection and economic development were addressed together (Purvis et al., 2019). From the 1970s onward, the concept of sustainability began to gain widespread attention, according to Purvis et al. (2019), and as noted by Caradonna (2014) the use in the literature increased rapidly during this period. In 1983, the UN created the World Commission on Environment and Development (WCED) to find ways to manage the growing environmental and development problems (Hedenus et al., 2022). This led to the report *Our Common Future* (WCED, 1987), which marked a key milestone in the spread and popularisation of sustainable development as a concept (Hedenus et al., 2022, Purvis et al., 2019). Although it has been criticised, for example, for placing disproportionate responsibility on poorer countries, it also established important links between previously separate areas and introduced the idea of justice both within and between generations (WCED, 1987, Hedenus et al., 2022).

In 1989, the organisation The Natural Step was founded, introducing a framework based on four sustainability principles (Caradonna, 2014). These principles were subsequently refined through the work of Holmberg and others (Holmberg, 1995, Azar et al., 1996). They describe conditions that must be fulfilled for a society to be sustainable. In 1992, the UN Conference on Environment and Development was held in Rio de Janeiro, resulting in the publication of the *Rio Declaration*, which comprised 27 principles, including the precautionary principle, to guide future sustainable development, and *Agenda 21*, a comprehensive action plan containing 120 initiatives (UN, 1993).

In 2000, the UN adopted the eight Millennium Development Goals (MDGs), to be achieved by the global community by 2015 (UN, 2016). The goals focused on reducing poverty and hunger, improving education and health, promoting gender equality, ensuring environmental sustainability, and strengthening global partnerships. While non-binding, these goals introduced measurable targets, which had previously been lacking. The Rio +20 UN conference on sustainable development in 2012 then initiated the process of developing a new set of global goals, which ultimately led to the 17 Sustainable Development Goals (SDGs) (UN, 2026). In connection with the Rio +20 conference, Griggs et al. (2013) argued in a Nature comment article that the SDGs must integrate planetary stability to reduce poverty and ensure human well-being. The article built on the MDGs and the planetary boundaries framework developed by Rockström et al. (2009), which defined a "safe operating space for humanity" based on limits for nine critical Earth-system processes. Based on this framework, Griggs et al. (2013) proposed six broad sustainable development goals.

Overall, sustainable development has evolved from an initial awareness of the environmental impacts of human activity to a broader understanding that environmental, economic and social aspects are all interconnected and must be addressed together, with the overarching goal of leaving no one behind (UN, 2024).

2.2.2 Perspectives and frameworks on sustainable development

The concept of sustainable development may in itself appear contradictory, since sustainable implies maintaining something over time, whereas development implies change (Hedenus et al., 2022). However, sustainable development can be understood as involving both preservation and transformation while balancing environmental, social, and economic considerations. A number of frameworks, definitions, and concepts have contributed to shaping the understanding of sustainable development. The following sections present the frameworks and concepts that form the basis of this thesis, particularly for the development of the definition of sustainable WWM in Chapter 4. These include the three dimensions of sustainability (Purvis et al., 2019), *Our Common Future* (WCED, 1987), the SDGs (UN, 2026), the planetary boundaries framework (Rockström et al., 2009), the Four Socio-Ecological Principles (Holmberg, 1995), and the concept of strong versus weak sustainability (Neumayer, 2013).

2.2.2.1 The three dimensions of sustainability

The three widely accepted dimensions of sustainability are the environmental, the social, and the economic. According to Purvis et al. (2019), there is no single original source, rather, the three-dimensions concept gradually evolved through several important contributions. Barbier's three-system model (biological, economic, and social) (Barbier, 1987), as well as the publications by Brown et al. (1987) and *Our Common Future* (WCED, 1987), all published in 1987, are commonly regarded as important early contributions to the three-dimensions perspective on sustainability.

The terms "the three pillars of sustainability", "the triple bottom line", and "people, planet and profit" are commonly used to describe the three dimensions of sustainability. While there is broad agreement that all three dimensions should be considered, views differ regarding how they relate to one another. Figure 2 shows three common perspectives: the overlapping circles model, which views sustainability as the intersection of the environmental, social, and economic dimensions (Barbier, 1987); the three pillars model, which represents sustainability as resting on three equal pillars (Purvis et al., 2019); and the nested model, in which the economy is embedded within society and society within the environment (Giddings et al., 2002), which was later further developed through the SDGs Wedding Cake model (Stockholm Resilience Centre et al., 2016).

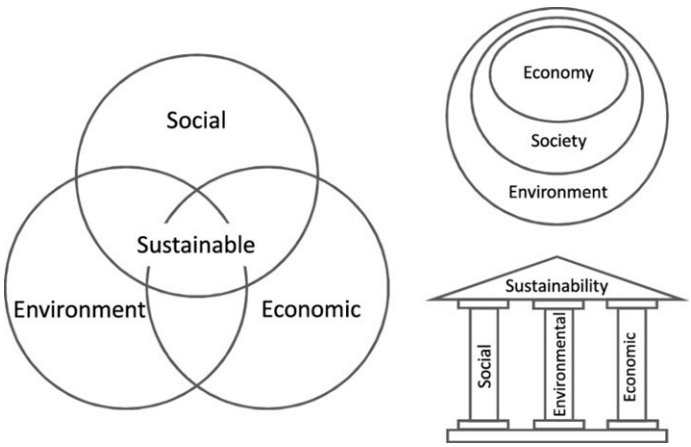


Figure 2. Different ways of illustrating how the three dimensions of sustainability, environmental, social, and economic, relate to one another, CC BY-ND 3.0. (Purvis et al., 2019)

2.2.2.2 Our Common Future

The *Our Common Future* Report, published by the WCED in 1987, is also referred to as the Brundtland Report, after Gro Harlem Brundtland, who chaired the WCED at the time. One of the report's most influential contributions is its definition of sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987). The report emphasised that economic development and sustainable development must be addressed together and expressed a need for both existing and new international and national organisations to support the transition towards more sustainable societies. According to the commission, these organisations would play a crucial role in influencing public awareness, attitudes, social values, and willingness to find and follow sustainable development pathways. Key areas highlighted in the report included food security, biodiversity and natural resources, energy, industry, and human settlements (WCED, 1987). These areas are well covered by the SDGs (UN, 2026), which can be seen as a continuation and expansion of *Our Common Future*, Agenda 21 (UN, 1993) and the Millennium Development Goals (UN, 2016).

2.2.2.3 The sustainable development goals

The SDGs constitute the main international framework for sustainable development today, having been adopted by all United Nations Member States in 2015. The SDGs were established with a time horizon of 2030 and consist of 17 goals (Figure 3), each with underlying targets and indicators. These targets specify the content of each goal, while the indicators facilitate the measurement of progress towards the targets and thereby the goals themselves (UN, 2026).

The SDG most directly related to the water sector is SDG 6, Clean Water and Sanitation. SDG 6 consists of eight targets and 11 indicators that cover safe drinking water, access to sanitation, pollution reduction and water reuse, water scarcity, water resources management, water ecosystems, cooperation between developed and developing countries in the water sector, and local water and sanitation management (UN, 2026). One particularly important target for WWM is SDG target 6.3, which aims to reduce the discharge of untreated wastewater and increase the proportion of wastewater that is safely treated. In 2022, only 76% of wastewater flows were reported to receive some form of treatment, based on data from 73 reporting countries (UN, 2024). Although SDG 6 is the most directly relevant goal for WWM, sustainable WWM is linked to and influenced by many other SDGs, as discussed in Chapter 4.

The EU has developed an approach for implementing the SDGs, which are integrated into many EU policies and strategies (EU, 2026b), including regulations on corporate sustainability reporting for large companies (EU, 2022). However, achieving the SDGs has become increasingly challenging due to growing geopolitical instability, armed conflicts, economic uncertainty, and security concerns, which may shift attention away from sustainable development (UN, 2025).



Figure 3. The 17 Sustainable Development Goals (UN, 2026). CC BY-ND 3.0.

2.2.2.4 The planetary boundaries

The planetary boundaries framework establishes limits for nine critical Earth-system processes. According to the framework, remaining within these boundaries reduces the risk of severe environmental change. The planetary boundaries are commonly illustrated as a spider chart, where each wedge represents one Earth-system process and a circle in the middle represents the safe operating space. The nine processes include climate change, biosphere integrity, biogeochemical flows (N and P), stratospheric ozone depletion, ocean acidification, freshwater change, land-system change, novel entities, and atmospheric aerosol loading. Since these processes interact with each other, crossing one boundary may also affect others, highlighting the need to consider them together (Rockström et al., 2009). The framework has been updated several times, most recently in 2025, when the assessment showed that seven of the nine boundaries had been transgressed (Sakschewski, 2025). Since September 2024, the Potsdam Institute for Climate Impact Research has published annual updates under the name *Planetary Health Check* (PIK, 2024).

This framework may at first appear relevant only to the environmental dimension of sustainability. However, if society and the economy are viewed as embedded within and dependent on environmental systems (see Figure 2), the framework becomes highly relevant to sustainable development. The planetary boundaries framework has been incorporated into later work, such as Griggs et al. (2013), Sustainable development goals for people and planet, which may in turn have influenced the development of the SDGs (UN, 2026).

2.2.2.5 Four socio-ecological principles

The four socio-ecological principles introduced in Section 2.2.1 were developed from a systems perspective and focus on the underlying causes of unsustainability rather than on the environmental problems themselves. According to Grönlund (2014), many environmental problems can be understood as consequences of violating these principles, which highlights their relevance in sustainability discussions.

The four principles are formulated as follows (Holmberg, 1995, Azar et al., 1996):

- Substances from the lithosphere must not systematically increase in the ecosphere.
- Substances produced by society must not systematically increase in the ecosphere.
- The physical basis for the productivity and diversity of nature must not be systematically deteriorated.
- Resources must be used fairly and efficiently with respect to meeting human needs.

The first principle addresses the accumulation of substances such as metals and fossil carbon extracted from the Earth's crust, implying that extraction rates should be restricted, recycling of materials increased, and the use of scarce elements reduced. The second principle concerns synthetic substances and compounds produced by society that nature cannot sufficiently degrade or remove. Such substances will accumulate if their production is faster than their degradation or removal rate. The third principle emphasises that ecosystems and biodiversity must not be systematically degraded, and that resource extraction must not exceed regeneration rates or reduce ecosystem productivity and diversity. The fourth principle concerns fairness and efficiency in resource use to ensure that human needs can be met sustainably through fair distribution and efficient use of natural resources and ecosystem services.

2.2.2.6 Strong and weak sustainability

Strong and weak sustainability represent two contrasting perspectives on sustainable development, with the key distinction concerning the substitutability of capital, particularly natural capital. Weak sustainability assumes that natural capital may be substituted by human-made capital, provided that the same functions are maintained and the total stock of capital remains constant. Strong sustainability, in contrast, argues that natural capital cannot be substituted by human-made capital (Neumayer, 2013). Within strong sustainability, Neumayer (2013) identifies two different variants: maintaining natural capital in value terms or preserving the physical stock of natural capital. The first variant allows one form of natural capital to replace another as long as the total value of natural capital is maintained. The second stricter variant does not allow any substitutions and focuses on keeping the function of the natural capital intact, meaning that resources may only be extracted if extraction rates do not exceed regeneration rates.

Strong and weak sustainability can also be seen as endpoints on a continuum. In weak sustainability, technology and people's opinions can affect what is regarded as substitutable. Luthman et al. (2022) propose a further subdivision into four categories: very weak, weak, strong, and very strong sustainability. Weak sustainability then differs from very weak by acknowledging that there are environmental thresholds, although these are still considered flexible. Strong sustainability requires that human activities do not degrade ecosystems beyond their functions, resilience, and assimilative capacity, while very strong sustainability implies that ecosystems should not be negatively affected at all.

2.3 Sustainability assessment methods

Achieving sustainable development requires informed decisions and, consequently, methods to compare the sustainability of available alternatives (Cinelli et al., 2014). This section provides a brief overview of selected sustainability assessment methods and approaches. Because multicriteria analysis (MCA) is central to the methodology applied in this thesis, it is described in greater detail.

2.3.1 Overview of different methods and approaches

To support sustainable decision-making, a range of assessment methods and related approaches have been applied in different contexts to evaluate alternative solutions from a sustainability perspective. Depending on the decision context, different methods may be used individually or in combination. Commonly used methods include life cycle assessment (LCA) (Corominas et al., 2013), life cycle costing (LCC) (Rebitzer et al., 2003), environmental impact assessment (EIA) (Lattemann and Höpner, 2008), carbon footprint analysis (Gustavsson and Tumlin, 2013), and cost-benefit analysis (CBA) (Ohlin Saletti et al., 2026), among others. Ness et al. (2007) provide a categorization of sustainability assessment tools.

LCA is a method for quantifying the environmental impacts associated with a product, process, or service throughout its life cycle, from raw material extraction and production to use and disposal (cradle-to-grave), or with a cut-off before the use and end-of-life phases (cradle-to-gate) (Baumann and Tillman, 2004). Typical impact categories in LCAs of WWT include climate change, acidification, and eutrophication (Corominas et al., 2013). These impacts can also be weighted and aggregated to facilitate comparison between alternatives (Baumann and Tillman, 2004). Regarding **LCC**, several definitions and methodological variants exist (Arias et al., 2025). Unlike traditional investment analysis, LCC considers costs throughout the life cycle of a product or system, but it does not traditionally include all environmental costs (Gluch and Baumann, 2004). Incorporating a circular economy perspective and indirect environmental costs into LCC provides a more comprehensive assessment of life cycle costs (Arias et al., 2025).

EIA is a systematic process for evaluating the likely environmental impacts of proposed projects before a decision is made on whether they should proceed (Jay et al., 2007). In the EU, EIA is regulated by the Environmental Impact Assessment Directive (EU, 2026a). While EIA considers a broad range of environmental impacts, **carbon footprint analysis** focuses specifically on climate impact by quantifying the total amount of direct and indirect greenhouse gas (GHG) emissions and removals within the defined system boundary. In WWM, this may refer to the GHG emissions associated with a WWTP (Gustavsson and Tumlin, 2013). Although conceptually similar to the climate change impact category in LCA, carbon footprint analysis may differ from LCA in terms of system boundaries and functional units. It may be applied in combination with cost assessments when the primary objective is to reduce climate impacts. **CBA** compares the costs and benefits of alternative solutions from a societal perspective by expressing them in monetary terms (Ohlin Saletti et al., 2026). The underlying principle is that an alternative is considered worthwhile if its total monetised benefits exceed its total monetised costs (Johansson and Kriström, 2018). CBA has also been incorporated into an MCA framework to assess the economic sustainability of alternative solutions (Söderqvist et al., 2015).

Each of these methods has its own strengths and limitations. Conventional LCA focuses on the environmental dimension and can be complemented by LCC to address economic aspects. Similarly, EIA focuses on environmental impacts, while carbon footprint analysis is limited to climate impact. CBA may be constrained by the difficulty of assigning monetary values to environmental and social impacts.

2.3.2 Multicriteria analysis

MCA, also referred to as multicriteria decision analysis (MCDA), is a decision-support method used to compare alternatives based on multiple, sometimes conflicting, criteria (Belton and Stewart, 2002). The basic elements of an MCA are a set of feasible alternatives, two or more criteria, and one or more decision-makers, and the method enables the selection, ranking, or sorting of

alternatives (Figueira et al., 2005). MCA has been regarded as a suitable method for sustainability assessment because of its flexibility and its ability to facilitate dialogue between stakeholders, analysts, and scientists (Cinelli et al., 2014). While many sustainability assessment methods focus on specific sustainability dimensions, MCA can consider environmental, economic, and social criteria simultaneously (Cinelli et al., 2014).

Belton and Stewart (2002) describe the MCA process in the phases of problem structuring, including defining the decision context and objectives; model building, which involves specifying alternatives, criteria, and performance values; model use, where the performance of the alternatives is evaluated and aggregated; and developing an action plan, in which the results are used to support decision-making. Sensitivity analysis is often included to evaluate how changes in assumptions, input data, or decision-makers' preferences affect the outcome (Belton and Stewart, 2002). Although MCA encompasses a wide range of methods (Rosén et al., 2015), a typical scoring- and weighting-based MCA process often includes (1) defining the aim and decision context; (2) specifying the alternatives; (3) setting the scoring scale; (4) selecting the criteria; (5) data acquisition; (6) scoring each alternative against each criterion; (7) assigning weights to the criteria; (8) calculating overall scores and (9) analysing and reflecting on the results. The order of these steps may vary slightly depending on the MCA variant. An MCA may be quantitative, qualitative, or semi-quantitative (Rosén et al., 2015), depending on the data and how performance is translated into scores. As alternatives are scored against each criterion, a scoring scale is required. The scale may be quantitative or qualitative and should include at least two defined reference points (Belton and Stewart, 2002). Scoring means translating the performance of each alternative for a given criterion into a score according to the defined scale, based on data, expert judgement, or other relevant information. Weighting, in contrast, should reflect the decision-makers' views on how strongly each criterion should influence the overall result in the specific case. The weights therefore represent value judgements about the relative importance of performance changes in different criteria.

There are numerous MCA variants that differ in their scoring, weighting, aggregation, and overall methodological structure with varying complexity, making a comprehensive review of all of them beyond the scope of this thesis. Therefore, a few representative approaches are briefly described below. The simplest and most widely applied methods are based on additive aggregation and are considered the easiest to explain to and understand by decision-makers (Belton and Stewart, 2002). In these methods, the score for each criterion is multiplied by its corresponding weight, and the weighted scores are summed to obtain an overall score for each alternative (Belton and Stewart, 2002). Multi-Attribute Utility Theory (MAUT) can use the same additive principle while incorporating utility functions to account for uncertainty, risk attitudes, and decision-makers' preferences (Kain and Söderberg, 2008, Belton and Stewart, 2002). For example, utility functions can be used to represent situations where the same improvement in performance is valued differently depending on where it occurs on the scale. Consequently, the relationship between performance and utility is not necessarily linear (Belton and Stewart, 2002).

In the Analytic Hierarchy Process (AHP), criterion weights and alternative scores are derived from pairwise comparisons of criteria and alternatives, respectively. Once the criterion weights and alternative scores have been derived, the overall score of each alternative can be calculated using a linear additive model (Linkov et al., 2006, Cinelli et al., 2014). In outranking methods, alternatives are compared pairwise to determine whether there is sufficient evidence that one alternative is at least as good as another when all criteria are considered (Belton and Stewart, 2002). Outranking approaches acknowledge that it is not always possible to identify a single best

alternative (Linkov et al., 2006). Common examples of outranking methods include ELECTRE, PROMETHEE, REGIME, and NAIADE (Kain and Söderberg, 2008, Vidal et al., 2019, Lai et al., 2008).

MCA has been widely applied and developed in the water sector due to its usefulness in addressing many types of complex decision problems (Malmqvist and Palmquist, 2005, Johannesdottir et al., 2021, Plakas et al., 2015, Lai et al., 2008, Radinja et al., 2019). Likewise, considerable work has been devoted to developing and proposing criteria and indicators suitable for WWM (Balkema et al., 2002, Malmqvist et al., 2006, Palme and Tillman, 2009, Cossio et al., 2020, Grönlund, 2019).

3 Future scenarios for wastewater management

In this thesis, the concept of the future is a key component. It is therefore necessary to clarify what is meant by the future, including both the time horizon and the conditions it entails.

As discussed in Sections 1.1 and 2.1.4, the wastewater sector is currently facing many challenges and uncertainties. Major and long-lasting infrastructure investments are needed in many locations due to factors such as ageing infrastructure, population growth, and stricter regulations (Wang et al., 2020, Lienert et al., 2015, Braga et al., 2026b). Since such projects often take a long time to plan and implement (Lienert et al., 2015), decisions must be made today despite considerable uncertainty about future conditions that may significantly affect WWM. These include future wastewater flows and pollutant loadings, regulations, precipitation patterns, and economic conditions.

Therefore, the aim of this study is to identify future conditions for WWM through scenario analysis, capturing a wide range of aspects and sustainability priorities to support well-informed planning of future wastewater systems (Section 1.2).

More specifically, this study proposes an approach to address the following objectives:

1. Identify future conditions and sustainability priorities relevant to WWM.
2. Estimate the likely development of these conditions as well as a range of possible outcomes.
3. Integrate the future conditions into coherent scenarios and demonstrate, through a case study, how they can be used to identify potential needs for future system upgrades.
4. Engage stakeholders to support robust and well-informed long-term planning.

This study, presented in Paper I, is based on a project carried out at Gryaab between 2017 and 2019. Gryaab owns and operates the Rya WWTP in Gothenburg, described in Section 2.1.3. The project engaged a broad range of participants and was led by a small core group responsible for planning and driving the work forward. As the approach was developed and applied within a real case, some elements were context specific. In this thesis, the aim is to highlight the overall approach, key steps, and general considerations to enhance the transferability and broader applicability of the methodology and its insights.

3.1 Approach to develop future scenarios for wastewater management

This section describes the methods applied in this study. First, the proposed stepwise approach, developed to address the study's objectives, is presented. Second, the tools used in the different steps of the approach are described.

An important starting point is to identify the central issue or problem, which helps to focus the analysis (Peterson et al., 2003). Although this may already be the primary motivation for conducting the scenario analysis, explicitly stating it helps to clarify the purpose and scope. Establishing the time horizon at an early stage is also fundamental, as it provides participants with a common reference point and supports coherent scenario development (Lienert et al., 2006).

The following steps outline the proposed approach.

3.1.1 Step 1. Local stakeholder assessment of future conditions

Step 1 establishes the foundation for the scenario development process by identifying the conditions expected to have a major influence on future WWM and by conducting an initial local assessment of these conditions.

The initial set of conditions can, for instance, be derived from existing frameworks and relevant literature and should include relevant regulatory, societal, environmental, economic, and technical aspects. In this study, an initial set of conditions was identified and selected by the project's core group, which possessed both technical expertise and local knowledge. These conditions were subsequently refined and assessed together with local stakeholders and covered the following main categories:

- Flows and population
- Treatment efficiency: regulatory requirements and/or ambitions
- Resource management
- Circularity
- Energy and chemicals
- Economic and societal priorities
- Other services to society (e.g. biogas, external sludge, slurry)
- Impacts on the surroundings
- Organisation, ownership, and technical coordination
- Capacity use (e.g. raw water protection, population growth, industrial services, stormwater)

The local stakeholder assessment was conducted through three workshops involving staff from the WWTP, as well as representatives responsible for water, WWM, and urban planning from municipalities in the region. The workshops had different themes (Table 2) and combined presentations, interactive group discussions, and individual responses. The pre-selected future conditions were discussed, and their potential future development was assessed through a combination of documented group discussions and the collection of individual responses. An important feature of the stakeholder process was the integration of results and opinions expressed at one workshop into the material prepared for the subsequent workshop.

The individual responses consisted of answers to statements and questions related to the future conditions and aligned with each workshop theme (Table 2). Based on the distribution of these responses, they were categorised as areas of consensus on (low response variability), a majority believe in (moderate variability with a clear majority) and disagreement on (distinct response clusters). Engaging stakeholders with diverse perspectives in this way shares similarities with the charrette methodology (Roggema, 2014) and is also recommended in scenario analysis (Peterson et al., 2003, Lienert et al., 2006).

Table 2. Themes and main discussion topics from the three stakeholder workshops conducted in Step 1 of the approach.

Workshop 1 "The future of the WWTP and its environmental impact"	The workshop's purpose was to introduce the project and its participants, and to discuss key topics such as regulatory requirements, future ambitions and resource management in wastewater treatment, including biogas, energy, and chemicals.
Workshop 2 "Sustainable water use and urban drainage: The role of the WWTP"	Key topics included the WWTP's impact on its surroundings (e.g., the urban landscape, traffic, odour, and noise), organisational and governance structures, system choices related to new connections and regional expansion, as well as wastewater flows and population dynamics.
Workshop 3 "Priorities and challenges for a sustainable society: The role of the WWTP"	The workshop covered economic considerations and competing societal priorities, sludge management and nutrient recycling, the allocation of treatment capacity (e.g., for environmental protection, drinking water protection and industrial services) and the provision of additional services to society.

3.1.2 Step 2. Initial formulation of future scenarios

In Step 2, relevant future conditions are combined into scenarios. The assessment of future conditions in Step 1 provides the basis for developing these scenarios. Two contrasting scenarios are developed for a defined future year, representing two plausible extremes within the range of possible outcomes. The actual future is assumed to lie somewhere within this range (Dominguez et al., 2009). While Step 1 may cover a broad range of future conditions, the scenario development at this stage focuses primarily on conditions that influence flows and loads to the WWTP. Current system conditions may be used as a baseline to facilitate the quantification of the scenarios.

For the two future scenarios for the case study, estimates of future flows were based on participants' responses from Step 1, combined with current data for the case study and a hydraulic flow estimation model developed for the region (Johnson et al., 2021). The model incorporated population projections, climate change impacts on flows, and anticipated changes to the configuration and operation of the sewer system, informed by the responses from Step 1. Projections of pollutant loadings (N, P, and BOD_7) within the scenarios were based on historical trends.

The two resulting future scenarios were used as input to a process model of the case study WWTP, supplemented by calculations performed in Microsoft Excel. The purpose was to illustrate how future scenarios may translate into operational consequences for an existing system. This included assessing the system's capacity to handle the scenarios, identifying potential upgrade needs, and comparing the results with anticipated regulatory requirements and other expectations regarding future WWM identified in Step 1.

3.1.3 Step 3. Expert-based adjustment of future scenarios

In Step 3, the future scenarios developed in Step 2 are adjusted and refined together with experts with diverse and relevant expertise, focusing on the conditions expected to have the greatest impact. This step aims to improve the plausibility of the scenarios for the chosen time horizon. The expert groups review the scenarios and provide input, and the outcome of this final step is a set of refined and validated future scenarios together with identified sustainability priorities.

Three expert groups were involved in the study (Swe1, Swe2 and Nordic). The composition of the first group (Swe1) was predefined and included experts from areas such as regulation, sewer systems, and wastewater treatment. This group participated in a one-day workshop that included presentations of the previous project work and the process modelling results for the initial scenarios developed in Step 2. The workshop also addressed other relevant topics, such as general trends in the Swedish and European wastewater sector and trends in the scientific literature related to sustainable WWM. These topics were intended to support knowledge exchange and stimulate discussion among the participants.

Participants provided individual responses regarding conditions related to future flows and loadings, regulatory requirements and future demands relative to current conditions. The participants were further divided into subgroups based on their areas of expertise. Within these subgroups, they provided quantified adjustments to the future conditions within their respective domains, by specifying minimum, maximum, and most likely values.

The two quantified future scenarios were then refined based on the adjustments proposed by the Swe1 group, and the resulting scenarios were again used as input to the process model of the case study WWTP, supplemented by calculations performed in Excel.

The second (Swe2) and third expert groups (Nordic) participated through a network meeting and a conference, respectively. Both groups were given a brief presentation of the project and were then asked to provide individual responses regarding conditions related to flows and loadings as well as regulatory requirements and demands for future WWM. The same questions were answered by participants in all three expert groups (Swe1, Swe2, and Nordic). In addition, the Swe2 and Nordic groups answered a question regarding sustainability priorities for WWTPs.

3.1.4 Tools for data collection, process modelling, and data analysis

To collect the individual responses to questions and statements during the workshops in Step 1 and Step 3 of the approach, a web-based digital tool for polling and participant interaction was used (Mentimeter, 2026). Participants' responses were given relative to the current situation. For questions in which participants ranked four alternatives, responses were normalised by assigning 4 points to rank 1, 3 points to rank 2, and so forth, after which the points assigned to each alternative were divided by the total number of points.

The quantified future scenarios were simulated using a dynamic process model developed in GPS-X 7.0 (Hydromantis, Hamilton, ON, Canada). The purpose was to evaluate the ability of the case-study WWTP to handle these potential future loadings. Similar approaches have previously been used to assess the impacts of future scenarios on WWTPs (Andersson et al., 2020, Arnell et al., 2017). The biological model used within the modelling software was Mantis, an extended version of Activated Sludge Model No. 1 (ASM1) (Henze et al., 2006), which includes nitrogen and organic matter processes. The process model represented the water line of the Rya WWTP (Section 2.1.3), i.e. not the sludge line. The screening and filtration steps were excluded from the model and instead represented through separate calculations. The model included primary settling, activated sludge treatment, trickling filters, secondary settling, and MBBRs. It was calibrated and validated using one year of historical daily operational data representing a typical year and was primarily used to predict effluent nitrogen concentrations for the two future scenarios.

For the removal of P and BOD_7 at the case study WWTP, this was calculated in Microsoft Excel. Inputs to both the process model and the Excel calculations consisted of daily average concentrations of BOD_7, N, and P, together with daily wastewater flows, over a one-year period.

These values were based on the population, specific loading and flow data in the quantified future scenarios for 2050. Daily flow variations were derived from a high-flow year, estimated to occur once every ten years. Future flows were estimated using a hydraulic flow model developed by DHI, which incorporated future rainfall series, climate change impacts, and sewer system modifications specified in the scenarios(Johnson et al., 2021).

The results from the process modelling and the Excel calculations were presented as annual average effluent concentrations (mg/L) and average daily mass flows (kg/day) for total nitrogen, total phosphorus, and total BOD_7.

3.2 The developed future scenarios and their implications

A result of this study is the proposed approach itself. When applied, the approach supports the development of plausible future scenarios for WWM, based on conditions expected to have a significant impact. The resulting scenarios can be used in calculations and process modelling to assess the implications of future conditions for the WWTP and identify potential upgrade needs. Furthermore, as is often recommended, stakeholders are involved throughout the process (Zheng et al., 2016), which can support long-term planning and decision-making.

The central issue and main objective of the case study was to establish and assess future conditions in order to provide a foundation for long-term expansion strategies for the existing WWTP. The time horizon was set to 2030–2070, with a primary focus on 2050.

3.2.1 Quantified future scenarios

From the local assessment in step 1 of the approach, the participants' individual responses were synthesised and are presented in Table 3. The assessed conditions were categorised into regulations and demands, external factors, and stormwater and separation of the sewer and grouped according to levels of agreement among the participants' responses.

The initial quantified future scenarios were based on the results from the local assessment (Step 1) together with current system conditions. The scenarios focused on population size, specific loadings of N, P, and BOD_7, daily flows and modifications to the sewer system. They were subsequently adjusted through input from the Swe1 expert group (in Step 3), and the resulting scenarios are presented in Table 4. The future scenarios were developed as two contrasting possible futures for 2050. However, both scenarios assumed the same population growth based on combined local, regional, and national predictions. The specific load of N per person was set lower than current levels in Scenario 1, motivated by potentially lower meat consumption and/or the implementation of source-separating systems in parts of the catchment area. However, the experts did not expect major changes related to source-separating systems in the urban context of Gothenburg. In contrast, in Scenario 2, the specific loads of both N and P were set slightly higher than current levels, which could instead be explained by higher meat consumption or other unforeseen developments.

The experts in the Swe1 group anticipated N to be a valuable asset in the future, where the energy consumption could influence where and how the recovery will take place (van der Hoek et al., 2018). They anticipated that P loads in wastewater most likely would decrease, as it is a limited resource. Regarding the specific load of organic matter (BOD_7), the Swe1 group assumed it to be at current levels or slightly higher. In general, the experts anticipated that wastewater flow per person would decrease due to lower potable water consumption and/or increased internal greywater recycling. The two scenarios differ in the rate of sewer system improvements, resulting in different levels of infiltration and inflow. In Scenario 1, a faster implementation rate of sewer

system improvements was set, while the rate was set lower in Scenario 2. The improvements were assumed to reduce flows as expected. Climate impacts were represented through climate factors affecting different flow types, where Scenario 1 reflected conditions similar to the current situation, while Scenario 2 assumed greater effects of climate impacts.

Table 3. Local stakeholder groups' views on future wastewater management conditions, grouped according to the three response categories: "Consensus on, A majority believe in, and Disagreement on" from Step 1 of the approach.

	Consensus on	A majority believe in	Disagreement on
Regulations and demands	Stricter discharge limits for BOD_7 and N New regulation for the removal of pharmaceuticals New regulation on P recovery	Stricter discharge limits for P Mandatory recovery of organic fertiliser from wastewater	New regulation requiring the removal of viruses, bacteria and other new micropollutants Mandatory recovery of N and micronutrients from wastewater
External factors	Odour control at the WWTP Prioritisation of the WWTP's visual appearance	Covered WWTP basins Multifunctional use of covered WWTP basins	Reduced noise from the WWTP Restriction on heavy traffic coupled to the WWTP
Storm-water	Sea flood barriers not implemented by 2070	Stormwater runoff management implemented	Climate-resilient cloudburst management plan implemented
Separation of the sewer system	Separate urine collection not implemented by 2070	Separate black water collection not implemented Local recycling of grey water implemented Reduced potable water consumption	Fully separated sewer system implemented Local heat recovery from wastewater upstream of the WWTP implemented

Table 4. Two quantified future scenarios representing plausible but contrasting future conditions, developed from the local assessment in Step 1, formulated in Step 2 and subsequently refined in Step 3. The scenarios include population size, specific loads, wastewater flows, sewer system modifications, and climate change impacts. Current system conditions (year 2017) are included for comparison.

	2050 Scenario 1	**2050 Scenario 2**	**2017 Conditions**
Population (persons)	1 100 313	1 100 313	763 064
Specific loads (g/p.d)			
N	11	14	13.6
P	1.3	1.8	1.7
BOD_7	80	100	84
Flows			
Mean daily flow (m3/s)	4.5	5.9	4.4
Domestic wastewater flow (L/p.d)	140	190	170
Improvements to the combined sewer system			
Conversion to separated system	20%	7%	-
Surface stormwater retention	17%	1%	-
Improvements to the separate sewer system			
Correction of incorrect connections	27%	6%	-
Repair or relining of pipes	36%	10%	-
Climate change impacts			
Leakage and drainage	As today	As today	-
Peak flows	As today	Higher than today	-

3.2.2 Anticipated future conditions by the expert groups

The results of the individual responses are illustrated using spider diagrams, showing anticipated future trends in flows and loadings (Figure 4) and regulations and demands (Figure 5). The anticipations regarding future flows and loadings were generally close to 5, representing conditions similar to those of today. Lower water consumption and increased impacts of climate change were conditions that, for all three groups, deviated most clearly from current levels. The expert groups also anticipated that future treatment requirements and demands would include stricter limits for N, P, and BOD_7, as well as new regulations on the removal of pharmaceutical residues, increased resource recovery, and reduced climate and environmental impacts, with chemical and energy use mentioned specifically. Sustainability priorities identified by the Swe2 and Nordic expert groups are presented in Figure 6. The highest sustainability priorities for future WWTPs were reduced resource use, lower energy consumption, and reduced climate impact, followed by low effluent pollutant emissions according to the Swe2 and Nordic expert groups. Low operational cost was given the lowest priority by both groups.

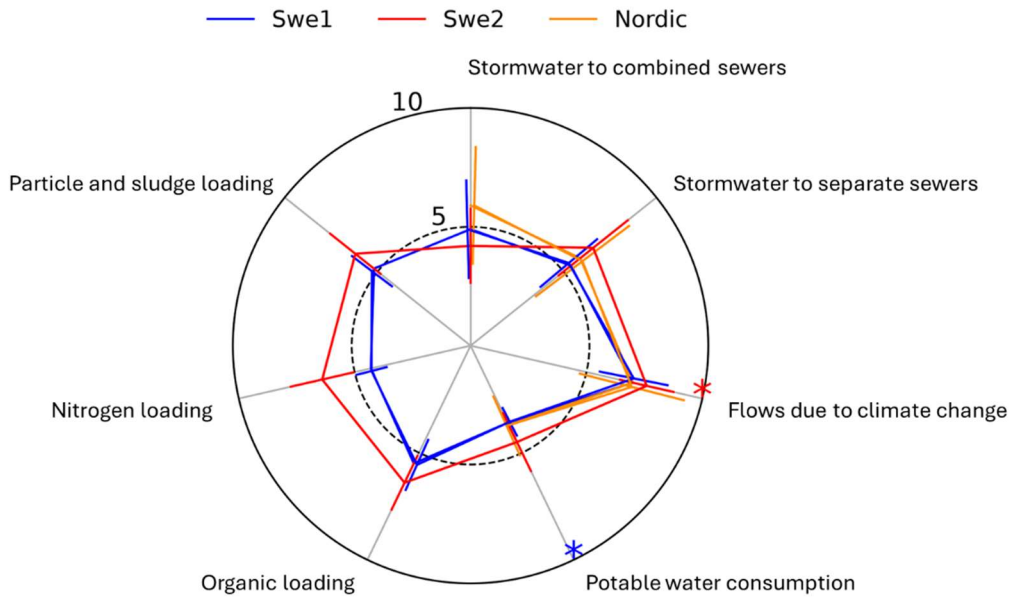


Figure 4. Anticipated future flow and load conditions according to the three expert groups: Swe1, Swe2 and Nordic. The scale ranges from 1 (centre point) to 10 (solid black line), with 5 (dashed line) representing current conditions. Values above 5 indicate higher flows or loads than current levels. The coloured lines connect the mean values for each expert group, while the error bars indicate the standard deviation for each variable. An asterisk (*) indicates a mean score significantly different from 5 for at least one of the groups ($p < 0.05$, one-sample t-test).

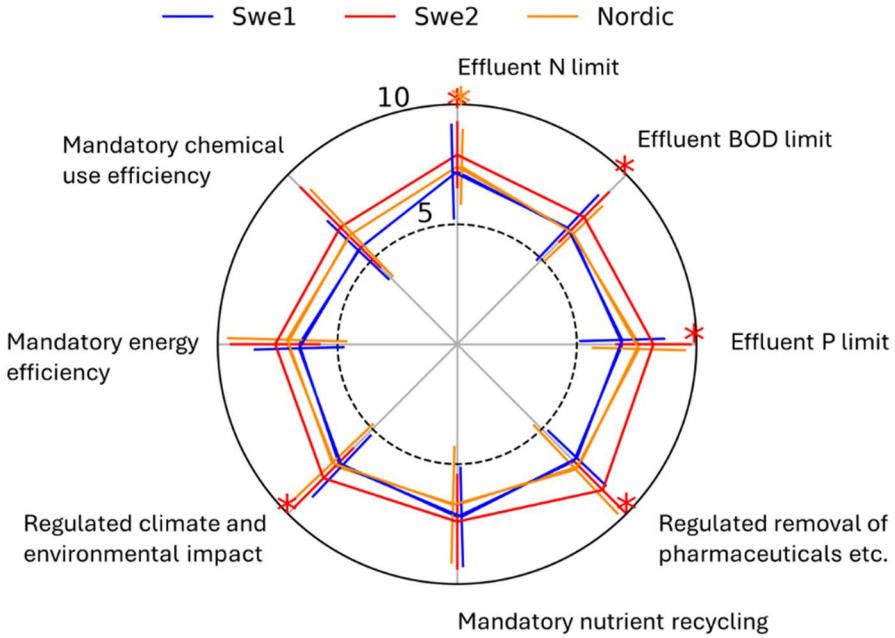


Figure 5. Anticipated regulations and demands according to the three expert groups: Swe1, Swe2 and Nordic. The scale ranges from 1 (centre point) to 10 (solid black line), with 5 (dashed line) representing current conditions. Values above 5 indicate stricter regulations or higher demands than today. The coloured lines connect the mean values for each expert group, while the error bars indicate the standard deviation for each variable. An asterisk (*) indicates a mean score significantly different from 5 for at least one of the groups ($p < 0.05$, one-sample t-test).

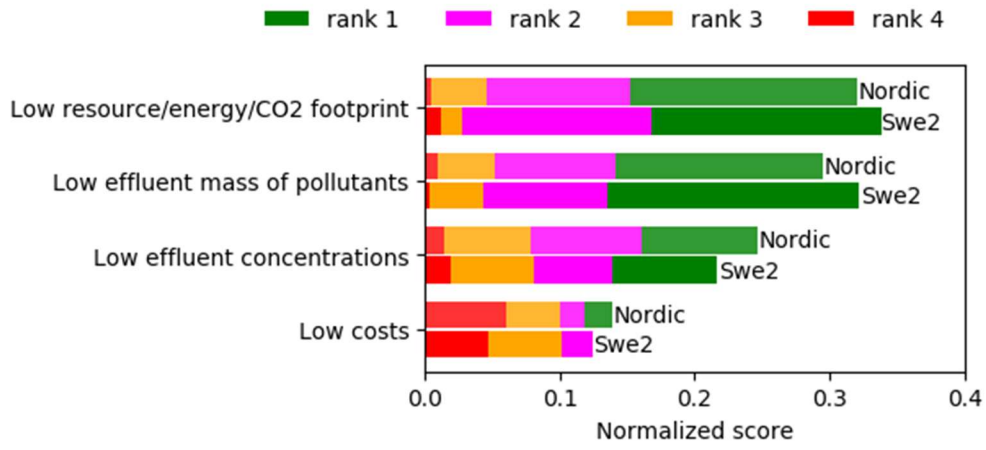


Figure 6. Ranking of sustainability priorities for future WWTPs by the Swe2 and Nordic expert groups. The bars show normalised scores derived from individual responses to: "For sustainable WWTPs, what is most important?". Higher scores indicate higher priority. Colours indicate the proportion of the total score contributed by each ranking (rank 1–4).

3.2.3 Consequences of the future scenarios for the case study WWTP

Table 5 presents a summary of the results from the dynamic process model simulations (for N) and the Excel calculations (for P and BOD_7) of the two scenarios (Table 4). The results show the simulated annual mean effluent concentrations and mass flows together with current discharge limits and future limits anticipated by the Swe1 group. The simulations showed that the total nitrogen concentration increased from 7.1 mg/L in Scenario 1 to 11.8 mg/L in Scenario 2, while the total BOD_7 concentration increased from 8.2 to 11.7 mg/L. Total phosphorus concentrations remained relatively similar between the scenarios, at 0.22 and 0.25 mg/L, respectively. The mass flows for all pollutants increased substantially from Scenario 1 to Scenario 2. All together the results showed that the existing WWTP would not be able to meet the anticipated discharge limits under either scenario without upgrades.

Table 5. Simulated annual mean effluent concentrations and mass flows for the two future scenarios for the case study WWTP in 2050. Discharge limits in effect in 2019 and predicted ranges for 2050, with the most likely values in parentheses, are included for comparison.

Parameter	Simulated scenario 1	Simulated scenario 2	Discharge limits (2019)	Predicted discharge limits (2050)[a]
Total N (mg/L)	7.1	11.8	8	2-6 (5)
Total P (mg/L)	0.22	0.25	0.3	0.05-0.2 (0.1)
Total BOD_7 (mg/L)	8.2	11.7	10	4-8 (6)
Total mass of N (kg/d)	2700	5900	-	-
Total mass of P (kg/d)	85	126	-	-
Total mass of BOD_7 (kg/d)	3200	5900	-	-

[a] Predicted range with the most likely value in parenthesis.
[b] This discharge limit is expressed as a three-year rolling average.

3.3 Reflections on the approach and the future scenarios

3.3.1 The role of local and expert knowledge in the proposed approach

In the proposed approach, the initial set of future conditions can be identified in various ways, such as through literature reviews or expert brainstorming. However, relying solely on such methods may fail to capture the local context. Engaging stakeholders, particularly those with

direct local knowledge or responsibility for the system in question, is recommended (Lienert et al., 2013), regardless of how the initial set of conditions is established. This helps ensure that the identified conditions are both broadly relevant and locally significant. A practical approach, applicable in many contexts, is to begin with a broad set of conditions identified from previous future scenario studies and then, in collaboration with stakeholders, iteratively refine and select those expected to have significant impacts on WWM in the local context.

In this case study, the initial selection of conditions was carried out by the project's core group, after which local stakeholders contributed to assessing their future development. The core group possessed both WWM expertise and local knowledge, providing a strong basis for identifying relevant future conditions. Such knowledge includes an understanding of how factors such as lifestyle and consumption patterns, weather and climate conditions, and existing wastewater infrastructure influence wastewater composition and flow patterns. It also includes insight into how regulatory frameworks, resource availability, geopolitical instability, emerging technologies, and the characteristics of the receiving environment may affect future WWM. Participants in the local assessment workshops and the Swe1 expert group also had the opportunity to provide input on the selection of conditions, although this role could have been more clearly defined and formalised as a distinct step in the approach. Translating the qualitative assessment of future conditions from Step 1 into quantified scenarios in Step 2 also requires substantial expertise and understanding of the specific wastewater system.

A key strength of the proposed approach is the involvement of stakeholders and different experts at multiple stages, which supports the validation of future conditions and scenarios and increases their robustness.

3.3.2 Considerations and transferability of the approach and results

Developing a range of scenarios, rather than a single "most likely" one, enables the formulation of strategies that can adapt to diverse future outcomes. Regarding the process modelling in Step 2, the simulations showed the Swe1 expert group the potential impacts of the future scenarios on the existing WWTP. However, process modelling is not necessarily required at this stage of the approach. Instead, it could be carried out after the scenario adjustments in Step 3, where the results could help demonstrate future upgrade needs to decision-makers at the WWTP. In this study, the dynamic process modelling simulations served primarily to illustrate the potential application and usefulness of the quantified future scenarios. Nevertheless, quantified future scenarios can be developed and provide valuable insights even without process modelling.

The benefits of stakeholder involvement must also be considered in relation to the resources required. In this project, stakeholder engagement was a key objective that was successfully achieved through the workshops. Representatives from the municipalities connected to the WWTP, including professionals in water management and urban planning, were involved throughout the process, improving scenario quality, building a shared understanding, and strengthening the basis for future decision-making (Seifert et al., 2019, Harris-Lovett et al., 2019).

Although the future scenarios presented in this study were developed for a specific case study, they may also have broader relevance. Some of the identified conditions are applicable to other contexts as well. For example, anticipated regulatory requirements may also be relevant in other regions within the same country or within the EU. In addition, the main categories of future conditions outlined in Section 3.1.1, as well as the specific conditions identified, may be useful in future scenario analyses in other contexts.

The anticipated sustainability priorities were ranked by expert groups with limited local knowledge (Swe2 and Nordic) and should therefore be interpreted as reflecting a general rather than a context-specific perspective. The expert groups assumed that regulatory requirements would be met. Furthermore, because the individual responses in the Nordic expert group were linked to participants' nationality and professional background, the data also provide opportunities for further analyses of how responses vary across national and professional groups, and how such differences may relate to regulatory frameworks in different countries. However, such analyses were beyond the scope of this thesis.

3.3.3 Comparison of anticipated and actual developments since 2019

As the original study was completed in 2019, subsequent developments provide an opportunity to reflect on some of the anticipated future conditions. Although the time horizons of 2030 and 2050 have not yet been reached, some trends can already be observed. For example, the anticipated introduction of regulatory requirements for pharmaceutical residue removal has since been reflected in the revised Urban Wastewater Treatment Directive (EU, 2024). There is also an increasing focus on improved energy efficiency and reduced climate impact in the revised Urban Wastewater Treatment Directive, which introduces increased requirements for monitoring greenhouse gas emissions (EU, 2024). The case study also anticipated stricter regulation of currently regulated pollutants. Consistent with this expectation, stricter discharge limits for N, P, and BOD_7 have been proposed in a new permit for the case study WWTP, although the permit has not yet been finalised. No regulatory developments related to mandatory chemical use efficiency or nutrient recycling have come to the author's attention to date. Changes in flows and pollutant loads have not yet been systematically compared with the scenario assumptions and are therefore more difficult to evaluate.

3.4 Summary and main findings

A stepwise approach was developed to identify future conditions relevant to WWM, combine them into future scenarios, and assess their implications for the case study, Rya WWTP. The approach included workshops with local stakeholders to identify and initially assess conditions expected to have a major influence on future WWM, followed by the formulation of two quantified scenarios representing contrasting but plausible future developments in terms of wastewater flows and loads, and then expert workshops to adjust them. The scenarios were simulated using a process model of the WWTP. The simulations showed that both scenarios would require upgrades to meet anticipated future discharge limits at the WWTP.

Many of the anticipated future flow and loading conditions were similar to those of today. Reduced water consumption and increased impacts of climate change showed clear deviations from current conditions. The anticipated future developments included stricter limits for N, P, and BOD_7, new requirements for the removal of pharmaceutical residues and increased resource recovery, as well as requirements for reduced climate and environmental impacts. The highest sustainability priorities for future WWTPs were reduced resource use, lower energy consumption, and reduced climate impact.

The proposed approach provides a structured way to develop future scenarios to support long-term decision-making in WWM. Combining local stakeholder knowledge with perspectives from WWM experts can help capture both locally relevant conditions and developments of broader importance to WWM. Evaluating the resulting scenarios using process models can help identify future system needs and support planning under uncertainty.

4 Definition of sustainable wastewater management

In Chapter 3, the concept of the future was addressed. Another key concept in this thesis, sustainable WWM, also requires clarification. Although its meaning may seem obvious, this is not necessarily the case. This chapter therefore explores how sustainable WWM can be defined, based on the same study as Paper II.

The challenges facing the wastewater sector have been highlighted several times (in Sections 1.1, 2.1.4 and Chapter 3), with the transition towards a more sustainable wastewater sector representing a central issue. As the deadline for achieving the SDGs approaches (UN, 2026), this transition becomes increasingly important for society, including the wastewater sector. Many scientific studies present new technologies and treatment concepts as sustainable solutions for the wastewater sector (Jönsson et al., 2024), but the underlying concept of sustainability on which such claims are based is often missing, as illustrated by studies such as those by Yang et al. (2017) and Lory et al. (2025).

Another common approach is to improve sustainability by mitigating an obviously unsustainable practice (Grönlund, 2014), such as high energy consumption or greenhouse gas emissions. However, focusing on a single issue without adopting a holistic perspective may result in trade-offs and unintended consequences. For instance, co-digestion with external organic waste may improve the energy balance of a WWTP through increased biogas production (Sarpong and Gude, 2021), but may also reduce the nutrient recycling potential if contaminants are introduced with the external organic waste. Such examples illustrate the need for holistic sustainability assessments that consider environmental, social, and economic dimensions. Furthermore, different sustainability concepts may lead to different conclusions about what is considered a sustainable solution. The distinction between strong and weak sustainability reflects differing views on whether natural capital can be substituted by human-made capital (Neumayer, 2013). Without an explicit definition, solutions based on these conflicting principles may both be regarded as sustainable.

Despite the growing focus on sustainability in WWM, there is still no commonly accepted definition of sustainable WWM, and although there is some consensus on the general understanding of sustainability, there is no consensus on how sustainability should be assessed in practice (Grönlund, 2014). The difficulty of establishing a unified definition may reflect differences in regional conditions, politics, technological development, economic constraints, and institutional capacities. Rather than emphasising these differences, the focus should be on identifying common sustainability principles that can guide WWM across contexts while allowing for local adaptation.

The aim of this study is to “provide a basis for sustainability assessment by defining sustainable WWM through a clear and detailed specification, grounded in scientific literature and established sustainability frameworks” (Section 1.2). The resulting definition includes criteria intended to support its practical application in sustainability assessments and decision-making. This supports the overall aim of the thesis by providing a foundation for assessing and comparing alternative pathways for future WWM from a sustainability perspective.

4.1 Definition development approach

To develop a definition of sustainable WWM, the following procedure was used:

1. Selection of well-established holistic sustainability frameworks and identification of their key elements.
2. Interpretation of each sustainability framework in the context of WWM.
3. Integration of framework interpretations to develop sustainability areas and objectives.
4. Identification of literature-based criteria relevant to the sustainability objectives.

Firstly (1), well-established and general frameworks and definitions of sustainable development were identified. Selection was based on their broad recognition and general acceptance in both the scientific literature and society at large. They either have a holistic perspective on sustainability or provide particularly important perspectives on specific sustainability aspects. The selection was not intended to be comprehensive, but rather to provide a broad and solid basis for defining sustainable WWM. Together, they cover important, sometimes overlapping, and sometimes complementary aspects of sustainability. The content of each framework or definition was summarised to capture its essential elements, structure, and underlying principles.

Secondly (2), each summary was interpreted in the context of WWM. With knowledge and experience from WWM, the sustainability statements, requirements, principles, and goals within the frameworks and definitions were translated into their implications for WWM.

Thirdly (3), the framework interpretations were synthesised and integrated to identify points of alignment, aspects coming from individual frameworks and potential divergences across frameworks. Using an inductive coding approach (Bingham, 2023), codes emerged directly from the analysis of the framework interpretations. The initial codes were refined through comparisons across the interpretations and overlapping and recurring codes were merged. This process ultimately resulted in eleven broader sustainability areas, each with a set of objectives that define its content and scope. The objectives reflect different aspects within each sustainability area, sometimes originating from different framework interpretations. Contradictions between or within the framework interpretations are discussed in Section 4.3.

Finally (4), the scientific literature was reviewed to identify sustainability criteria and indicators for the wastewater sector. Based on this, criteria were systematically aligned with the defined sustainability areas and objectives. The literature-based criteria provide additional justification for their inclusion and facilitate the practical application of the proposed definition through sustainability assessments, such as MCA.

4.2 Proposed definition

The sustainability frameworks and concepts introduced in Section 2.2.2 form the basis for developing a definition of sustainable WWM. These include the three dimensions of sustainability (Purvis et al., 2019), *Our Common Future* (WCED, 1987), the SDGs (UN, 2026), the planetary boundaries framework (Rockström et al., 2009), the Four Socio-Ecological Principles (Holmberg, 1995), and the concept of strong versus weak sustainability (Neumayer, 2013).

4.2.1 Sustainability areas and objectives

Eleven sustainability areas, each comprising one or more objectives, are proposed to collectively define sustainable WWM (Figure 7). The areas are named according to their main focus, which in some cases is an action (e.g. nutrient recovery) and in others a sustainability challenge (e.g.

climate change). The following sections describe each area and its objectives, together with their connections to the underlying sustainability frameworks. Table 6 provides an overview of how the frameworks and concepts relate to the eleven sustainability areas.



Figure 7. The eleven areas that together constitute the proposed definition of sustainable wastewater management.

4.2.2 Sanitation and hygiene

The first sustainability area is Sanitation and hygiene, which is directly connected to SDG6 (UN, 2026). Ensuring access to adequate sanitation for all is recognized as a human right and must therefore be a non-negotiable part of sustainable WWM. WWM is essential for preventing exposure to pathogens. Maintaining a strong focus on hygiene is therefore important both when establishing new wastewater systems and when modifying existing ones.

Objective 1.1: Provide safe and adequate sanitation solutions that minimize the risk of waterborne infection and health issues.

4.2.3 Ecosystems of the recipients

The second sustainability area is Ecosystems of the recipients. The area is clearly supported by the SDGs (UN, 2026), *Our Common Future* (WCED, 1987) and the planetary boundaries framework (Rockström et al., 2009), all emphasizing ecosystem protection and reduced environmental impact.

Several SDGs are directly relevant to this area, particularly SDG targets 6.3, 6.6, and SDG14. SDG target 6.3 focuses on improving water quality through reducing pollution and the release of untreated wastewater, while SDG target 6.6 concerns the protection and restoration of water-related ecosystems. SDG14, Life below water, includes target 14.1, which aims to prevent and significantly reduce marine pollution of all kinds, including from land-based activities such as nutrient pollution from wastewater. The planetary boundary relating to biosphere integrity clearly states that loss of biodiversity and extinctions must be avoided, while the boundary for biogeochemical flows of N and P implies that discharges of N and P from wastewater systems must be kept at environmentally safe levels, recognizing that such levels may vary depending on geographical and ecological conditions.

Our Common Future can be interpreted as implying that wastewater discharges should not cause long-term negative effects on receiving ecosystems, since future ecological conditions and human needs connected to these ecosystems cannot be fully predicted.

Objective 2.1: Avoid the release of untreated wastewater to the receiving ecosystems.

Objective 2.2: Treat all wastewater to site-specific pollutant levels that do not negatively affect receiving ecosystems in the short or long term. Relevant pollutants may include nitrogen, phosphorus, and organic matter.

Objective 2.3: Contribute to the restoration of damaged water-related ecosystems.

Objective 2.4: Locate discharge points for treated wastewater to minimize impacts on receiving ecosystems.

4.2.4 Recovery of nutrients

The third sustainability area is Recovery of nutrients, an area influenced by the planetary boundaries framework (Rockström et al., 2009) and the Socio-ecological principles (Holmberg, 1995).

The Planetary Boundary concerning biogeochemical flows of N and P addresses human disruption of the natural nutrient cycles of these essential elements and emphasises that human additions of them to the biosphere must be limited (Rockström et al., 2009). Wastewater contains substantial amounts of both N and P and in modern WWM systems, most of the P ends up in the sludge, while the recovery of N is generally limited with large amounts instead being converted to nitrogen gas during treatment.

The fourth socio-ecological principle, concerning fair and efficient use of resources to meet human needs, is particularly relevant for P, a finite and critical resource for fertiliser production and food security. From this perspective, sustainable WWM should promote increased recovery of P to preserve it for future generations. Recovery of other nutrients and organic compounds is also important to reduce resource losses and support long-term societal needs.

Objective 3.1: Maximize the recovery of finite resources, such as phosphorus, to preserve these resources for future generations.

Objective 3.2: Ensure high recovery of non-finite resources, such as nitrogen and organic compounds, for soil improvement and other societal needs.

4.2.5 Protection of biodiversity and ecosystems

The fourth sustainability area is Protection of biodiversity and ecosystems, referring to ecosystems beyond the recipients. This area is supported by the planetary boundaries framework (Rockström et al., 2009) as well as *Our Common Future* (WCED, 1987), the SDGs (UN, 2026) and the socio-ecological principles (Holmberg, 1995).

The planetary boundary concerning land-system change can be negatively affected by the expansion of wastewater systems. Regarding the planetary boundary of biosphere integrity, ecosystems where wastewater systems are located, and potentially areas where sewage sludge is applied, may be negatively impacted. The third socio-ecological principle also states that physical manipulation or disturbance of ecosystems can impact long-term productivity and biodiversity. SDG15, concerning ecosystems on land, is in line with the planetary boundaries and the socio-ecological principles. The loss of species and genetic resources was also one focus area in *Our*

Common Future. Although the establishment of a wastewater system negatively affects nature and may involve irreversible physical changes, it is also necessary for protecting ecosystems in waters and on land.

An indirect consequence of WWM may be the cultivation of biomass for renewable energy or chemical production. This production must be carried out in a way that preserves biodiversity and the long-term productivity.

Objective 4.1: Locate and design WWM systems and system expansions to minimize disturbances and negative impacts on ecosystems.

Objective 4.2: Produce crops for chemicals or renewable energy in ways that preserve biodiversity and the long-term productivity of the land.

Objective 4.3: Ensure that sludge application does not negatively affect biodiversity or cause accumulation of harmful substances in the soil.

4.2.6 Substances of emerging concern

The fifth sustainability area is Substances of emerging concern, which is addressed by several frameworks, including the planetary boundaries (Rockström et al., 2009), the SDGs (UN, 2026) and the Socio-ecological principles (Holmberg, 1995).

The Planetary Boundary of novel entities and the SDG target 6.3 both emphasise reducing pollution and minimising the release of hazardous chemicals. According to DANVA (2018), achieving SDG14 also requires removal of pharmaceutical residues and plastics, including microplastics. Furthermore, the second Socio-ecological principle concerns substances produced by society that must not systematically accumulate in the ecosphere.

As WWTPs represent one pathway through which these substances spread further into the environment, this provides a potential opportunity for their separation and removal. However, achieving a long-term sustainable solution through the phase-out of such chemicals requires addressing their sources through bans, substitution, and source reduction.

Objective 5.1: Avoid the use of hazardous substances within WWM.

Objective 5.2: Influence upstream stakeholders to minimize the use of hazardous substances.

Objective 5.3: Contribute to minimising hazardous substances in the environment through appropriate wastewater treatment.

4.2.7 Climate change

The sixth sustainability area is Climate change, which is directly linked to the planetary boundaries framework (Rockström et al., 2009), the SDGs (UN, 2026) and the Socio-ecological principles (Holmberg, 1995).

Both the planetary boundaries concerning climate change and ocean acidification have CO_2 emissions as a common underlying driver. The first socio-ecological principle further implies that use of fossil fuels for chemicals, electricity, and other purposes must be phased out. Mitigating climate impact is also important for achieving several SDGs, and SDG13 specifically addresses both climate change mitigation and adaptation. Actions to mitigate climate change may also positively influence freshwater systems and the global hydrological cycle.

WWM can reduce its climate impact by increasing the use of renewable energy sources, minimising emissions of N_2O and CH_4 from wastewater treatment, sewer systems, and wastewater discharge, and recovering and efficiently utilizing carbon and other resources in wastewater. Climate change mitigation and adaptation clearly need to be considered in the planning of future WWM systems.

Objective 6.1: Use only renewable resources for chemicals, energy, and other purposes.

Objective 6.2: Minimize emissions of N_2O and CH_4 from WWM, including the sewer systems, treatment processes, and wastewater discharge.

Objective 6.3: Contribute to the reduction of CO_2 emissions from fossil sources in society, for instance by replacing fossil fuels with biogas from wastewater treatment.

Objective 6.4: Optimize utilisation of the existing carbon in the wastewater.

Objective 6.5: Establish WWM systems with flexibility to climate change and develop climate adaptation plans together with relevant stakeholders.

4.2.8 Energy and resources

The seventh sustainability area is Energy and resources, which is closely linked to the previous area, Climate change. Energy and resources has clear connections to *Our Common Future* (WCED, 1987), the SDGs (UN, 2026), and the socio-ecological principles (Holmberg, 1995).

Regarding the connection between climate change and energy, much of the energy use and electricity supply globally still originate from fossil fuels, and the combustion of fossil fuels accounts for a large share of global CO_2 emissions. Replacing fossil energy sources with renewable sources is therefore important. This is central to SDG7, which aims to ensure access to affordable, reliable, sustainable and modern energy for all, and is also highlighted in *Our Common Future*.

WWM systems require energy for electricity, heating, cooling, and chemical production, but also contain recoverable energy in the form of thermal and chemical energy. Optimizing energy use, recovery and efficiency is important to ensure that renewable energy resources are sufficient to meet societal needs.

From a sustainability perspective, energy is often regarded as a limited resource, and parallels can be drawn to the use of other resources, such as metals, scarce elements, and fossil raw materials. To align with the socio-ecological principles and to achieve SDG12, ensure sustainable consumption and production patterns, the use of resources should be minimized and recycling increased. Achieving SDG12 also requires minimising waste generation and encouraging more sustainable consumer behaviour.

Objective 7.1: Optimize the utilisation of the energy resources in wastewater, including thermal and chemical energy.

Objective 7.2: Minimize energy use and establish energy-efficient WWM systems.

Objective 7.3: Avoid treating wastewater to lower levels than required by receiving ecosystems if this results in increased energy or resource demand.

Objective 7.4: Contribute to the production of renewable energy.

Objective 7.5: Minimize the extraction and use of resources and contribute to recycling.

4.2.9 Water reuse

The eighth sustainability area is Water reuse, which is most directly supported by the SDGs (UN, 2026).

Within SDG12, Responsible consumption and production, consumer behaviour should be guided towards decreased water consumption. Increased water reuse also contributes to SDG target 6.4, improving water-use efficiency, and may also contribute to SDG7 through increased energy efficiency. Water reuse in WWM can be achieved through, for instance, greywater recycling (reuse of lightly contaminated household wastewater) and the reuse of treated wastewater for various applications. However, social acceptance remains an important challenge for some forms of wastewater reuse.

Objective 8.1: Contribute to reduced water consumption by influencing consumer behaviour.

Objective 8.2: Contribute to increased water reuse upstream, downstream, and within WWTPs.

4.2.10 Cooperation at different levels

The ninth sustainability area is Cooperation at different levels, an area that *Our Common Future* (WCED, 1987), the SDGs (UN, 2026), the planetary boundaries framework (Rockström et al., 2009), and the Socio-ecological principles (Holmberg, 1995) all emphasise as crucial for achieving sustainable development.

The SDGs provide many examples of the need for cooperation. According to SDG targets 6a and 6b, as well as SDG17, sustainable WWM should contribute through cooperation and knowledge sharing at both local and international levels. To support sustainable cities, as highlighted in SDG 11 and partly SDG9, the water sector needs to cooperate with other sectors. SDG target 6.5, concerning integrated water resources management, highlights the need for planning and cooperation among stakeholders. SDG4 further emphasises cooperation with educational and research institutions to support equitable education for all. Overall, cooperation and interaction are needed across sectors and at all levels, from local to international.

Objective 9.1: Cooperate and share knowledge at local, national, and international levels within the water sector and with other sectors.

Objective 9.2: Cooperate with stakeholders and educational and research institutions to contribute to equitable education for all.

4.2.11 Fairness and other social aspects

The tenth sustainability area is Fairness and other social aspects, which is strongly linked to the Socio-ecological principles (Holmberg, 1995), the SDGs (UN, 2026) and *Our Common Future* (WCED, 1987).

The fourth socio-ecological principle concerns fair and efficient use of resources to meet human needs. Similarly, Our Common Future emphasises meeting the needs of both present and future generations. This perspective is also reflected in SDG10, Reduced Inequalities, which addresses social and economic inequalities within societies. WWM can represent a fair use of resources within a society by providing equitable sanitation and wastewater services for everyone. However, resources also need to be fairly distributed globally, and major inequalities in access to adequate wastewater services remain. In relation to SDG10, increased investment and resources are therefore needed in many developing countries.

Many other SDGs also address social sustainability issues relevant to the wastewater sector. These include contributing to safe sanitation and public health (SDGs 1–3), improving gender equality (SDG5), promoting decent work (SDG8), and reducing inequalities (SDG10). Important efforts include ensuring equal opportunities, diversity, non-discrimination, and respect for labour rights. In relation to SDG11, Sustainable Cities and Communities, wastewater systems are a critical part of societal infrastructure and should therefore be developed to support fair, inclusive, and resilient societies.

Objective 10.1: Contribute to resources for WWM being fairly distributed globally.

Objective 10.2: Design and operate wastewater systems that minimize waste generation and resource consumption to support fair and efficient use of resources in society.

Objective 10.3: Contribute to a labour market characterized by gender equality, respect for labour rights, diversity and no discrimination or inequalities.

Objective 10.4: Contribute to fair, inclusive, and resilient societies.

4.2.12 Economy

The eleventh and final sustainability area is Economy, which is most directly addressed in *Our Common Future* (WCED, 1987) and the SDGs (UN, 2026).

The integration of economic development and sustainable development was one of the central messages in *Our Common Future*. Several SDGs are also strongly connected to the economic dimension, particularly SDG8 on sustainable economic growth, while SDG9, 10, and 12 also address important economic aspects for sustainable development and economic resources are also essential for achieving progress towards many of the other sustainability goals.

Economic considerations play a central role in WWM. As discussed in Sections 2.1.4 and Chapter 3, many investments are long-term and may affect economic conditions for decades, highlighting the importance of sustainable economic planning.

Objective 11.1: Make responsible long-term investments that support economic stability.

Objective 11.2: Contribute to resilient and well-functioning WWM infrastructure that supports society and economic development.

Objective 11.3: Establish WWM systems that are adaptable and flexible to future changes in loading, demands, and external conditions, thereby contributing to economic stability.

Table 6. Overview of how the sustainable development frameworks included in this study relate to the eleven sustainability areas of the proposed sustainable wastewater management definition.

	Our Common Future (WCED, 1987)	**SDGs** (UN, 2026)	**Planetary boundaries** (Rockström et al., 2009)	**Four socio-ecological principles** (Holmberg, 1995)	**Strong and weak sustainability** (Neumayer, 2013)
1. Sanitation and hygiene	X	X			
2. Ecosystems of the recipients	X	X	X		X
3. Recovery of nutrients			X	X	
4. Protection of biodiversity	X	X	X	X	X
5. Substances of emerging concern		X	X	X	
6. Climate impact		X	X	X	
7. Energy and resources	X	X	X	X	X
8. Water reuse		X	X		
9. Cooperations	X	X	X	X	
10. Fairness and social aspects	X	X		X	
11. Economy	X	X			

4.2.13 Sustainability criteria for the proposed definition

The proposed definition is complemented by sustainability criteria (Table 7), organized according to the same structure as the sustainability areas in the definition. It is also clarified which objective each criterion relates to. The criteria have been derived from the literature, and criteria from different sources addressing similar aspects using different terminology have been combined to avoid overlap.

According to Malmqvist et al. (2006), criteria should be selected based on the specific context and the total number should be kept low to facilitate efficient assessment processes and stakeholder involvement. Availability of information and stakeholder preferences are important when selecting criteria. Direct measurements are not always necessary, as proxy indicators may provide sufficiently accurate information. Criteria may also be refined into more detailed sub-criteria when needed. When applied, the criteria must be expressed in appropriate units, as exemplified in the referenced literature. Units may be adapted to the specific context or standardized (e.g., per population equivalent or wastewater volume) to improve comparability across assessments. Linking the criteria to parameters in the revised Urban Wastewater Treatment Directive (EU, 2024) could further strengthen the applicability of the proposed definition.

Sustainability criteria are typically categorized into the environmental, social, and economic dimensions. Malmqvist et al. (2006) suggested adding the dimensions “health and hygiene” and “functional and technical”, while Marques et al. (2015) suggested including “governance” and “infrastructure”. However, introducing additional dimensions may increase complexity. The key consideration is that relevant aspects are included, while the arrangement into dimensions is more of a structural issue. The aspects proposed by Malmqvist et al. (2006) and Marques et al. (2015) can therefore be incorporated within the three fundamental sustainability dimensions. For example, “health and hygiene” is most closely related to the social dimension and similarly, “functional and technical” and “infrastructure” largely concern economic aspects such as flexibility, adaptability, and reliability. The focus of the proposed definition is thus to ensure a comprehensive coverage of relevant sustainability aspects without introducing additional dimensions or explicitly defining relationships between them (Jenssen et al., 2007).

Table 7. Sustainability criteria for wastewater management, organized into the 11 areas of the proposed definition. Each criterion is linked to the corresponding objectives (in parenthesis) and supported by prior publications.

Sustainability areas	Associated criteria (corresponding objective)	Reference
1. Sanitation and Hygiene	Access to urban water services (1.1)	(Marques et al., 2015, Hellström et al., 2000)
	Health risks of infection or chemical risk (1.1)	(Malmqvist et al., 2006, Foxon et al., 2002)
2. Ecosystems of the recipients	Proportion of wastewater flows safely treated (2.1)	(UN, 2026)
	Recipient impact: Discharge of phosphorus, nitrogen, organic matter, suspended solids, pathogens, acidifying substances etc. (2.2, 2.3)	(Malmqvist et al., 2006, Marques et al., 2015, Machado et al., 2020, Jenssen et al., 2007, Balkema et al., 2002, Muga and Mihelcic, 2008, Foxon et al., 2002, Hellström et al., 2000)
	Effects on the aquatic ecosystem, exceeding environmental quality standards (2.2, 2.3, 2.4)	(Malmqvist et al., 2006)
3. Recovery of nutrients	Nutrient recovery: Recycling of N, P, K, S to agriculture or other applications (3.1, 3.2)	(Malmqvist et al., 2006, Jenssen et al., 2007, Balkema et al., 2002, Foxon et al., 2002)
4. Protection of biodiversity and ecosystems	Negative impact on groundwater (4.1, 4.2)	Rosén et al., 2015, Hellström et al., 2000)
	Air pollution: Polluting emissions to air excl. greenhouse gases (4.1, 4.2)	(Rosén et al., 2015, Foxon et al., 2002)
	Contamination of sediment (4.1)	(Rosén et al., 2015)
	Land area used (4.1, 4.2)	(Balkema et al., 2002, Foxon et al., 2002)
	Negative impact on soil ecosystems affecting biodiversity, land fertility or habitats, exceeding environmental quality standards (4.1, 4.2)	(Malmqvist et al., 2006, Rosén et al., 2015)
	Toxic compounds, heavy metals and substances of emerging concern to soil and agricultural land habitats, exceeding environmental quality standards (4.3)	(Malmqvist et al., 2006, Rosén et al., 2015, Jenssen et al., 2007, Foxon et al., 2002, Hellström et al., 2000)
5. Substances of emerging concern	Use of products with hazardous components (5.1)	(Jenssen et al., 2007)
	Chemical use (5.1)	(Balkema et al., 2002, Foxon et al., 2002)
	Release of heavy metals, toxic compounds and substances of emerging concern to water (5.2, 5.3)	(Balkema et al., 2002, Foxon et al., 2002)
6. Climate change	Climate impact through greenhouse gas emissions (6.1, 6.2, 6.3, 7.4)	(Malmqvist et al., 2006, Jenssen et al., 2007, Foxon et al., 2002, Hellström et al., 2000)
	Optimal resource utilisation or reuse (6.4)	(Foxon et al., 2002, Balkema et al., 2002)
7. Energy and resources	Energy use: Total energy use and energy recovered (7.1, 7.2, 7.3, 7.4)	(Malmqvist et al., 2006, Marques et al., 2015, Jenssen et al., 2007, Muga and Mihelcic, 2008, Balkema et al., 2002, Foxon et al., 2002, Hellström et al., 2000)
	Resource use: finite, critical non-renewable resources (7.5, 10.2)	(Malmqvist et al., 2006, Rosén et al., 2015, Jenssen et al., 2007, Balkema et al., 2002, Foxon et al., 2002, Hellström et al., 2000)
	Waste production, split in recyclable and non-recyclable waste (7.5, 10.2)	(Rosén et al., 2015, Balkema et al., 2002)

Sustainability areas	Associated criteria (corresponding objective)	Reference
8. Water reuse	Total drinking water consumption (8.1, 8.2)	(Foxon et al., 2002)
	Wastewater production (8.1, 8.2)	(Foxon et al., 2002)
	Water reuse: Percentage water reused (8.2)	(Foxon et al., 2002)
9. Cooperation at different levels	Knowledge and organisational requirements (9.1)	(Muga and Mihelcic, 2008, Balkema et al., 2002, Hellström et al., 2000)
	Local participation, responsibility, development and knowledge gain (9.1, 9.2)	(Malmqvist et al., 2006, Rosén et al., 2015, Marques et al., 2015, Muga and Mihelcic, 2008, Balkema et al., 2002, Foxon et al., 2002)
	Collaboration and communication between actors (9.1, 9.2)	(Malmqvist et al., 2006)
	External consequences: institutional, legal, political support (9.1, 9.2)	(Malmqvist et al., 2006, Balkema et al., 2002)
10. Fairness and other social aspects	Public perception: awareness, acceptance, understanding (10.3, 10.4)	(Malmqvist et al., 2006, Rosén et al., 2015, Foxon et al., 2002, Hellström et al., 2000)
	Social inclusion and contribution to equity (10.3)	(Rosén et al., 2015, Foxon et al., 2002)
	Work environment (10.3)	(Marques et al., 2015, Hellström et al., 2000)
	Safety and security risks (10.3)	(Jenssen et al., 2007)
	Effects on surroundings: Noise, smell, other disturbances or contribution to aesthetical or recreational value (10.1, 10.4)	(Rosén et al., 2015, Jenssen et al., 2007, Muga and Mihelcic, 2008, Balkema et al., 2002)
11. Economy	Annual cost for investment, operation, staff, transport, emission tariffs etc. (11.1)	(Malmqvist et al., 2006, Machado et al., 2020, Marques et al., 2015, Jenssen et al., 2007, Muga and Mihelcic, 2008)
	Financial risk (11.1)	(Malmqvist et al., 2006, Foxon et al., 2002)
	User cost (11.1)	(Muga and Mihelcic, 2008)
	Technical function: Reliability, work demand, maintenance needs, technical demand, performance level (11.2, 11.3)	(Malmqvist et al., 2006, Machado et al., 2020, Marques et al., 2015, Jenssen et al., 2007, Balkema et al., 2002, Foxon et al., 2002, Hellström et al., 2000)
	Vulnerability and robustness (11.2, 11.3)	(Balkema et al., 2002, Foxon et al., 2002, Hellström et al., 2000)
	Flexibility and adaptability (6.5, 11.3)	(Foxon et al., 2002, Hellström et al., 2000)

4.3 Implications of the proposed definition

The proposed definition provides a structured basis for understanding and assessing sustainable WWM. However, the proposed definition and the sustainability frameworks on which it is based also raise important questions regarding their implications and potential trade-offs. These aspects are discussed in this section.

One of the frameworks underlying the proposed definition is that of planetary boundaries. Previous studies have evaluated WWM solely from a planetary boundaries perspective.

For instance, Ryberg et al. (2021) assessed a Danish WWM company based on its allocated share of the framework's "safe operating space for humanity". They found that more than half of the allocated Planetary Boundary shares were exceeded, particularly those related to climate change and N and P flows. Similar results were reported by Lima et al. (2023), in which two WWM systems exceeded several planetary boundaries, including those for climate change and N and P flows. These findings suggest that current WWM systems face considerable challenges in operating within the safe operating space defined by the planetary boundaries framework, making alignment with the broader proposed definition even more demanding.

Strong and weak sustainability provide perspectives not only on the proposed definition but also on WWM more broadly. Wastewater system expansion may be acceptable according to weak sustainability, provided that the total capital stock is preserved. However, it is difficult to anticipate all present and future values associated with the existing capital. As a result, substituting natural capital with human-made capital cannot ensure that all of its values are preserved. For example, replacing natural swimming areas with concrete swimming pools, or wild fish stocks with fish farms, may preserve certain functions while still resulting in the loss of other ecological, social, or cultural values. At the same time, current centralized wastewater systems, and society more generally, do not align with strong sustainability principles, and even decentralized, source-separating, or wetland-based systems affect natural capital. Like all human infrastructure, wastewater systems inevitably influence land and receiving ecosystems, both positively and negatively. Sustainable WWM should strive to minimize negative impacts while maximizing positive ones. Further distinctions between weak and strong sustainability, as suggested by Luthman et al. (2022), may support practical application, with positions between weak and strong sustainability likely to be the most relevant depending on local conditions and priorities.

The proposed definition also gives rise to practical questions. For instance, it includes an objective within the sustainability area "Ecosystems of the recipients" stating that the release of untreated wastewater must be avoided. This raises the question of whether combined sewer overflows (CSOs), which are present in wastewater systems today, can be considered compatible with sustainable WWM, since they result in discharge of untreated wastewater. However, CSOs serve an important function by preventing sewer flooding in buildings and cannot be immediately eliminated. Sustainable WWM should therefore, in the long term, strive to implement solutions that avoid the release of untreated wastewater while ensuring that such discharges occur only under exceptional circumstances.

Another example concerns the area "Recovery of nutrients", which emphasises maximizing phosphorus recovery and ensuring high recovery of nitrogen and other nutrients. Wastewater contains substantial amounts of P, most of which, in modern WWTPs, ends up in sludge, while recovery of N at WWTPs is generally very limited. In Sweden, only 25% of the P in sludge was recycled to agriculture in 2017, and although this share may have increased since then, the reported share was far below the estimated potential of 90% (Lorick et al., 2021). Increasing nutrient recovery is therefore an important aim for sustainable WWM. Potential pathways include both innovative zero-discharge concepts (Pandey, 2025) and source-separating systems that differ from most established wastewater systems (Kjerstadius et al., 2015), although their implementation may involve significant technical and economic challenges. One pathway, established in some countries such as Sweden, is the application of sewage sludge on agricultural land to utilize recovered P and other nutrients. However, there are differing views regarding the potential environmental consequences of this practice. Manzetti et al. (2015) suggest that

biodiversity may be negatively affected. This relates to the area “Protection of biodiversity and ecosystems”, which requires that sludge application does not harm biodiversity or cause the accumulation of harmful substances in soils. Long-term studies of sludge application on land indicate only moderate accumulation of heavy metals such as copper, zinc, and mercury in soils (Börjesson and Kätterer, 2018). Other substances, such as aluminium and iron, used in precipitation chemicals, are naturally abundant in the environment, implying a relatively low risk of causing systematic accumulation there (Karlsson, 1997).

Achieving the resource-related objectives of the proposed definition requires reduced waste generation, efficient resource use, and more sustainable consumption patterns. Treatment processes should therefore be selected and operated to minimize waste production and resource consumption. This may become particularly important during periods of resource scarcity, when the availability of treatment chemicals may be limited.

The sustainability area “Energy and resources” emphasises how WWM can contribute more broadly to sustainability through renewable energy use and energy recovery, for example through biogas production, carbon utilisation, and heat recovery (Aleksić et al., 2023). At the same time, WWM is generally energy-intensive, particularly when aiming for very low effluent concentrations of pollutants. From an energy-efficiency perspective, wastewater should therefore not be treated beyond what is necessary to protect aquatic ecosystems. Similarly, CSOs and excessive infiltration and inflow reduce system efficiency, as they require the pumping and treatment of stormwater and groundwater. Improving energy efficiency is essential if renewable energy resources are to be sufficient to meet broader societal needs. However, the area “Substances of emerging concern” may require additional WWT technologies for the removal of pharmaceutical residues. Most technologies for pharmaceutical removal are energy- or carbon-intensive and implementing them, while simultaneously increasing energy efficiency in line with the sustainability area “Energy and resources”, presents a significant challenge. Likewise, the area “Ecosystems of the recipients” may require higher pollutant removal efficiencies, generally requiring technologies with higher energy and resource consumption. Ultimately, this raises questions of prioritisation: should the focus be placed on enhanced pollutant removal or on ensuring access to clean and sufficient energy? More broadly, this illustrates the challenge of balancing one sustainability objective against another. Future sustainable WWM may therefore need to focus not only on improved pollutant removal, but also on more energy- and resource-efficient treatment processes. Increased renewable energy production within WWM, together with broader developments in the energy sector, may help meet rising energy demands. In addition, fulfilling the objectives of the area “Water reuse”, through lower water consumption and increased water reuse may also be beneficial from an energy perspective.

The sustainability area “Economy” raises another type of discussion. Grönlund (2014), referring to Balkema et al. (2002), notes that economic sustainability is often interpreted as paying for itself, meaning that costs should not exceed benefits. However, Grönlund (2014) argues that if monetary flows are viewed as subsystems supporting the long-term sustainability of society, certain sectors may justifiably operate at higher costs. Food production, drinking water supply, and wastewater treatment can all be regarded as such sectors, since inadequate wastewater treatment may have severe consequences for society. Grönlund (2014) also argues that economic sustainability often focuses too narrowly on individual WWTPs, rather than considering an entire region.

The examples above illustrate only some of the trade-offs that arise when applying the proposed definition. Other trade-offs will inevitably remain among the sustainability objectives for WWM.

This is already evident within the SDGs, where multiple objectives are not easily compatible, and combining them with other sustainability frameworks and concepts further increases the complexity. A universal solution is unlikely to be suitable in all contexts, since regulatory conditions, system development, local priorities, and resource availability vary considerably between regions and countries. Consequently, each case should be assessed individually to identify the most sustainable pathway under the given conditions, as emphasised by Kehrein et al. (2020). This highlights the need for a sustainability definition that is sufficiently structured to support assessment, yet flexible enough to allow for contextual variation.

A circular perspective has been identified as a promising pathway towards sustainable WWM (Silva, 2022). Principles of the circular economy, such as reduction, recycling, and resource recovery, align closely with the proposed definition and are further supported by the framework presented by Kehrein et al. (2020). While Kehrein et al. (2020) provide valuable guidance for the strategic planning of water resource recovery facilities, the proposed definition places greater emphasis on the broader conceptual foundation of sustainable WWM and more strongly integrates the social dimension through areas such as “Sanitation and hygiene”, “Cooperation at different levels”, and “Fairness and other social aspects”. The proposed definition also relates well to the work by Estévez et al. (2025), who developed a roadmap and criteria for a potential environmental certification system for WWM. Shared elements include the protection of ecosystems and biodiversity, sustainable resource use, and considerations related to safety and labour conditions. However, while Estévez et al. (2025) primarily build on existing European legislation, the proposed definition adopts a more forward-looking perspective by defining requirements for future sustainable WWM. It also places greater emphasis on cooperation, knowledge sharing, and global fairness within the social dimension and may in that sense complement their work.

4.4 Summary and main findings

A holistic definition of sustainable WWM was developed by synthesising established sustainability frameworks and interpreting them from a WWM perspective. The resulting definition consists of eleven sustainability areas, each specified by objectives and complemented by literature-based criteria. The sustainability areas are sanitation and hygiene, ecosystems of the recipients, recovery of nutrients, protection of biodiversity and ecosystems, substances of emerging concern, climate change, energy and resources, water reuse, cooperation at different levels, fairness and other social aspects, and economy. The objectives specify what sustainable WWM should strive for within each area, while the criteria support the practical application of the definition.

The implications of the proposed definition and important trade-offs between its objectives were further examined in the chapter. These included whether CSOs, while important for managing high flows, are compatible with sustainable WWM given their associated pollutant discharges and energy use. The relation to other sustainability concepts was also considered. The definition generally aligns well with a circular economy perspective, while applying the principles of strong and weak sustainability to WWM presents practical challenges. For practical application of the definition, the objectives can guide criteria selection in assessment methods such as MCA. Trade-offs and differences in regulations, existing systems, and local priorities can then be considered in the assessment, allowing the definition to provide a common sustainability framework that can be adapted to different WWM contexts.

5 Sustainability assessment method for wastewater management

In Chapter 4, a definition of sustainable WWM was proposed. Building on this definition, this chapter presents the development of a method for assessing the sustainability of alternative pathways. The studies and methodological developments presented here build on the work reported in Papers III and IV.

5.1 Methodological foundation and objectives

A structured method is needed to assess the sustainability of WWM, and it should be flexible enough to accommodate different decision problems, contexts, and case-specific conditions and provide a broader evaluation than a final ranking of alternatives alone. Stakeholder participation is recognised as an important element of sustainability assessments (Bousquet et al., 2023, Huang et al., 2019, Kehrein et al., 2020, Rosén et al., 2015) and the method should therefore facilitate such participation. Based on the proposed definition of sustainable WWM (Chapter 4), sustainability criteria were compiled and presented in Section 4.2.13, providing a suitable basis for MCA. Although other assessment methods, including some presented in Section 2.3, could potentially serve as a methodological foundation, MCA meets many of the identified requirements. It enables quantitative, qualitative, and often conflicting criteria to be considered and integrated within the same framework (Marques et al., 2015, Johannesdottir et al., 2021, Radinja et al., 2019) and supports stakeholder involvement. These characteristics make MCA a suitable methodological foundation for this study.

However, MCA also has limitations, particularly regarding the transparency of the scoring, weighting and the results (Kaliszewski and Podkopaev, 2016, Lai et al., 2008). Subjective judgments may strongly influence results, and dominant individuals may disproportionately affect the outcome of the assessment (Norrman et al., 2020). Within MCA, the weighting stage is generally regarded as particularly susceptible to subjective influence.

One aim of this thesis is to develop a practical sustainability assessment method for WWM (Section 1.2). To achieve this, an MCA-based assessment method was developed, and the following objectives were established to guide its development:

1. Provide a practical method for the consistent comparison of alternatives in WWM.
2. Enable systematic and holistic assessment of relevant sustainability aspects, future conditions, and uncertainties.
3. Support the evaluation of impacts, trade-offs, and the robustness of the outcomes.
4. Improve transparency in scoring and weighting.
5. Test and refine the method through case studies based on real WWM decision issues.

5.2 The proposed sustainability assessment method

This section presents the proposed sustainability assessment method and its methodological developments, which build on the nine-step MCA process outlined in Section 2.3.2. An overview of the method is provided in Figure 8. The method is designed to accommodate different WWM decision problems, from organic micropollutant removal and sludge management to WWTP expansion and wastewater collection. System boundaries, sustainability criteria, and weights are therefore adapted to the specific decision context. For example, some social aspects may be

highly relevant when comparing household food waste collection systems but less important when selecting technology for a WWTP expansion.

To improve scoring and weighting, several methodological developments were introduced. A central development is the use of WSRs, introduced as a means of quantifying interrelations between different criteria. WSRs are described in detail in Section 5.2.8.1 and their iterative use during the assessment allows participants to better understand the implications of the assigned weights and the relative valuation of criteria. WSRs also enable comparisons with reference values from society and with other sustainability assessments. Additional developments include linking the scoring scale to future acceptable levels, performing scoring before weighting, checking the linearity of scoring for quantifiable criteria, and calculating the value per weighted score. Together with sensitivity analyses that separately favour each alternative, these developments aim to increase transparency while providing a clearer understanding of the robustness of the results and what most strongly influences them.

A weighting method based directly on WSRs is also proposed. In this approach, weights for quantitative criteria are derived from assigned monetary valuations using the WSR equation. In addition to increasing transparency, the method provides a practical procedure that may facilitate the weighting process. Literature-based reference values and monetary valuations from previous MCA studies are compiled to provide a basis for assigning weights and, where appropriate, anchoring them in established valuations.

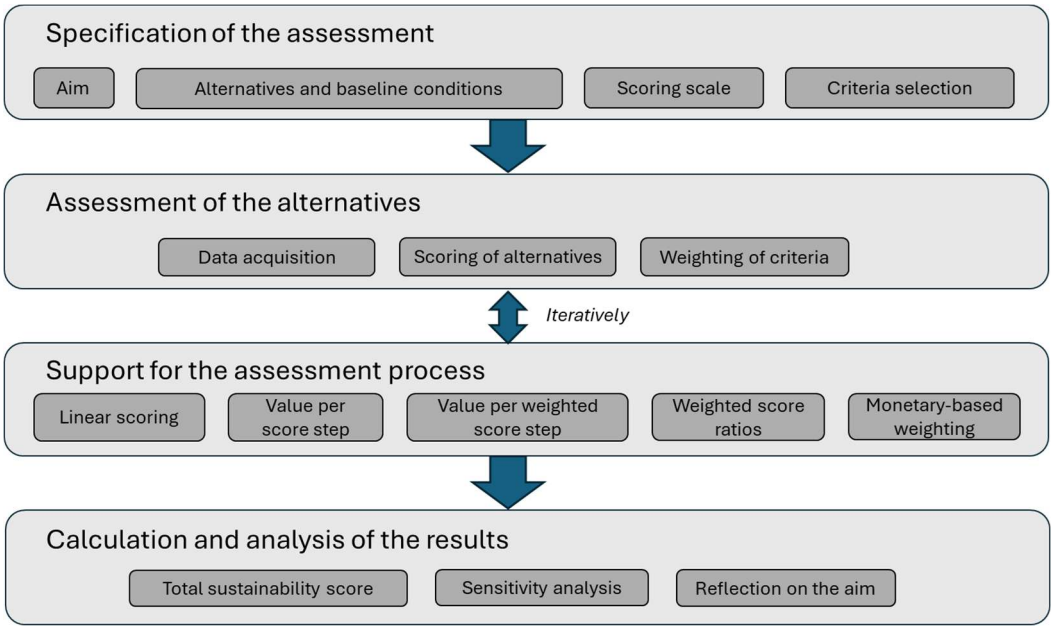


Figure 8. Overview of the proposed sustainability assessment method.

5.2.1 Assessment groups and roles

The assessment is carried out by one or more assessment groups, involving relevant stakeholders and led by a moderator, sometimes referred to as an analyst. It is advantageous to use separate assessment groups for data acquisition, scoring, and weighting. Since weighting reflects preferences and priorities, it is appropriate for the weighting group to include decision-makers. For scoring, it is beneficial to involve individuals with expertise relevant to the criteria and the issues being assessed.

5.2.2 Assessment aim

The aim of the assessment is formulated as clearly and precisely as possible, including whether it concerns present or future conditions by specifying a year or time period. In addition to identifying the most and least sustainable alternatives, the aim may also be to determine which aspects have the greatest influence on the results, as well as the robustness of the results under changing conditions.

5.2.3 Specification of included alternatives

The assessed alternatives are specified in sufficient detail, with all assumptions and underlying conditions clearly documented. This information should be available to both assessment participants and those evaluating or using the results afterwards. Examples of relevant conditions include geographical location, system boundaries, wastewater flows, treatment efficiency, and information on energy and chemicals. Some conditions may be identical across all alternatives, while others differ. Variations in baseline conditions can be examined through sensitivity analyses.

5.2.4 Scoring scale

The scoring scale is established by defining the number of score steps and their general interpretations. Whenever possible, these interpretations should be linked to current or future goals, regulations, or other established reference values or benchmarks at the organisational, local, or national level. In the proposed method, the scoring scale ranges from 1 to 5, with score 3 generally representing the future acceptable level for each criterion (Table 8). Scores are not restricted to integers and may include decimal values.

Table 8. The proposed scoring scale, linked to future acceptable level.

Score	**1**	**2**	**3**	**4**	**5**
Interpretation	Considerably worse than future acceptable level	Worse than future acceptable level	Future acceptable level	Better than future acceptable level	Considerably better than future acceptable level

5.2.5 Criteria selection

A relevant set of criteria from the environmental, social, and economic dimensions is selected for the assessment. To ensure a holistic sustainability assessment, all three dimensions should be represented. The criteria presented with the definition of sustainable WWM in Chapter 4, compiled based on previous sustainability assessments, can serve as a basis for criteria selection. For each selected criterion, relevant data, proxy data, or other supporting evidence should be available to provide a basis for scoring (Malmqvist et al., 2006).

5.2.6 Data acquisition

The data required for the assessment, including facts, numerical values, and other information needed to score each alternative against each selected criterion, are collected. The data should reflect the intended year or period of the assessment. Reference data, such as information from current operations or existing systems, may also be included to support the scoring process and provide a basis for comparison.

5.2.7 Scoring of alternatives

The participants in the scoring group score one criterion at a time for all alternatives based on the data presented by the moderator, using the defined scoring scale. Scores may also be assigned to variants included in the sensitivity analyses, either at this stage or later in the assessment process.

As part of an iterative process, the scores of quantifiable criteria are used to perform linearity checks and calculate the value per score step, as described below.

5.2.7.1 Linearity check of scoring

For criteria with a quantifiable data basis, the linearity of the scoring is checked. For each criterion, the underlying data are plotted against the assigned scores as the scoring is performed, with each alternative represented by one data point. When three or more data points are available, the linearity of the relationship can be checked, and scores adjusted to follow linearity.

5.2.7.2 Calculation of value per score step

For quantifiable criteria with linear scoring, the value corresponding to one score step is calculated, using Eq. 1.

$$V_i = \frac{\Delta v}{\Delta s} \qquad \textit{Eq. 1}$$

where V_i is value per score step for criterion i, Δv is the range in values (e.g., cost in € or energy in kWh) and Δs is the range in scores corresponding to those values.

5.2.8 Weighting of criteria

The participants in the weighting group assign weights to the criteria using an arbitrary scale from 1 to λ (e.g., λ = 20 or 100). The coefficients are automatically normalised to percentages with a total sum of 100%, allowing the group to focus on whether to give a criterion a high or a low priority, rather than exact percentage values. The assigned weights should reflect how strongly a one-score-step change in a criterion should influence the total sustainability score, rather than the general importance of the criterion itself (Marques et al., 2015). The group is also encouraged to define minimum and maximum weights for each criterion to be used in sensitivity analyses.

The weighting step may also include the calculation of weighted score ratios (WSRs), the value per weighted score step, comparisons of WSRs with reference values, and monetary-based weighting, as described below.

5.2.8.1 Calculation of weighted score ratios

WSRs are calculated based on the assigned weight of each criterion (out of 100%) and the fact that, for linearly scored quantifiable criteria, each score step corresponds to a known change in performance (e.g. in tonnes CO_2e, kWh, €, etc). WSRs express the relationship between criteria by comparing changes in performance that have an equivalent impact on the total sustainability score. WSRs are calculated using Eq. 2.

$$WSR\ [unit\ A/unit\ B] = \frac{V_A}{V_B} \cdot \frac{w_B}{w_A} \qquad \textit{Eq. 2}$$

where WSR is the weighted score ratio for criteria A and B, V_A and V_B are the values per score step (Eq. 1) and w_A and w_B the weights assigned to criteria A and B.

WSRs may be expressed in units such as €/tCO_2e, €/kWh, or kWh/tCO_2e. For interpretation and comparison with reference values, it is often practical to express all WSRs relative to cost. It is therefore recommended to start the weighting procedure by assigning and fixing the weight of the cost criterion before assigning weights to the remaining criteria. However, WSRs between other criteria are also possible, such as tCO_2e/kWh.

5.2.8.2 *Value per weighted score step*

For non-quantifiable criteria, value per weighted score step can be expressed in the units of any quantifiable criterion included in the assessment by using the relationship between criterion weights. For instance, work environment can be expressed as cost per weighted score step using Eq. 3, assuming annual cost is criterion C, and work environment is criterion D. This represents the corresponding annual cost for changing work environment by one score step. The value per weighted score step can also be used as guidance when assigning weights.

$$Value\ per\ weighted\ score\ step\ \ [unit\ C/weighted\ score\ step\ D] = V_C \cdot \frac{w_D}{w_C} \qquad \textit{Eq. 3}$$

where V_C is the value per score step for criterion C and w_D and w_C are the weights assigned to criteria D and C.

5.2.8.3 *Comparison of WSRs with reference values*

WSRs can be compared with reference values, expressed in the same units (e.g. €/tCO_2e, €/kWh). The comparison can be used to assess the plausibility of the assigned scores and weights, as well as to support adjustments of criterion weights, including the determination of minimum and maximum weights, as recommended in the proposed method. By relating the assessment to external reference values, the basis for weighting decisions can be clarified and their justification strengthened. Potential sources of reference values include local or national goals, valuations from authorities or other sectors of society, and previous sustainability assessments (see Section 5.3). Weights can also be derived directly from predefined reference values, as described in Section 5.2.8.4.

5.2.8.4 *Monetary-based weighting procedure*

Monetary-based weighting is an alternative procedure based on the WSR approach, provided that annual cost is included as one criterion. The procedure consists of the following steps:

- A weight, w_A, is assigned to the annual cost criterion. An arbitrary scale from 1 to λ (e.g., λ = 20 or 100) may be used to facilitate the procedure.
- A monetary valuation is determined for each quantifiable criterion i, expressed as a monetary value per criterion unit. Reference values may be used to support this process (Section 5.3).
- A monetary valuation is determined for each non-quantifiable criterion. This valuation can be interpreted as the corresponding additional annual cost to improve the non-quantifiable criterion by one score step.
- Criterion weights, w_i, are calculated using Eq. 4:

$$w_i[unit] = MV_i \cdot w_A \cdot \frac{V_i}{V_A} \qquad \textit{Eq. 4}$$

where w_i is the weight of criterion i, MV_i is the assigned monetary valuation of criterion i, w_A is the weight assigned to the annual cost criterion, and V_i and V_A are the values per score step for criterion i and annual cost (Eq. 1), respectively. For non-quantifiable criteria, V_i is 1.

- All criterion weights are normalised using Eq. 5 so that their sum equals 100%.

$$w_i' = \frac{w_i}{\sum_{k=1}^{m} w_k} \cdot 100 \qquad \textit{Eq. 5}$$

where w_i' is the normalised weight (in %) of criterion i, w_i is the original weight of criterion i and $\sum_{k=1}^{m} w_k$ is the sum of all original weights across the m criteria.

5.2.9 Calculation of total sustainability scores

The resulting total sustainability score for each alternative is calculated using a simple additive model with the assigned criterion weights and corresponding scores of the alternatives, according to Eq. 6 (Belton and Stewart, 2002):

$$S_j = \sum_{i=1}^{m} \frac{w_i'}{100} \cdot s_{ij} \qquad \textit{Eq. 6}$$

where S_j is the total sustainability score for alternative j, m the number of included criteria, w_i' the normalised weight of criterion i (in %) and s_{ij} the score assigned to alternative j for criterion i.

5.2.9.1 Sensitivity analyses

Several types of sensitivity analyses can be performed. For sensitivity analyses involving changes to individual scores or weights, the resulting total sustainability scores are recalculated using Eq. 6 with the modified values. A sensitivity analysis particularly useful for assessing the robustness of the results is combined weighting, in which the strengths of each alternative are favoured, one alternative at a time. This analysis requires that minimum and maximum weights have been defined for all criteria. For each alternative, a "best for alternative X" scenario is created by assigning the maximum weight to criteria where the alternative has received a high score (4 or 5), the minimum weight to criteria where the alternative has received a low score (1 or 2) and retaining the original weight for criteria with a middle score (3). The resulting set of weights is then normalised and used to calculate new total sustainability scores for all alternatives, using Eq. 6. This type of sensitivity analysis illustrates how high the sustainability score of an alternative could become if all weights were adjusted in its favour, while also clarifying whether any combination of weights within the defined ranges could change the ranking of the alternatives.

5.2.10 Analysis and reflection

In the final step of the proposed method, the total sustainability scores are interpreted in relation to the aim of the assessment. Major and minor contributors to the total sustainability scores may be analysed for each alternative, together with the results of sensitivity analyses. The alternatives are ranked according to their total sustainability scores, and sensitivity-analysis results may be considered when interpreting the overall sustainability ranking. This provides an indication of the robustness of the ranking of the original alternatives. This phase can also help identify potential knowledge gaps and priorities for further investigation. If the sensitivity analyses indicate that certain criteria strongly influence the results while also being associated with substantial uncertainty, this may motivate further investigation. Conversely, criteria that have little or no influence on the results may be given less attention in similar future projects.

5.3 Reference values for monetary valuation of criteria

The proposed sustainability assessment method in Section 5.2 can include the calculation of WSRs (Section 5.2.8.1) or monetary-based weighting (Section 5.2.8.4). Monetary reference values can support and help justify weighting decisions in both approaches. This section presents a compilation of reference values for climate impact, energy use, and discharges of nitrogen, phosphorus, and BOD to receiving waters. The aim was to identify reliable and relevant reference values representing a sufficiently broad range. Sources included scientific publications, industry guidelines, policies and other documents commonly used in practice, including those from Swedish government authorities and the EU. Sources were selected based on their relevance, established use, and traceability. For climate impact, many suitable sources were available,

whereas for other criteria, fewer sources were available. Values derived from previous MCA studies reflect how criteria were implicitly valued, even though the corresponding WSRs were not explicitly calculated in the original studies.

To ensure consistency and comparability, all values were first adjusted for inflation in their respective currencies (SCB, 2026, U.S. Bureau Of Labor Statistics, 2026, Bank of Canada, 2026, Eurostat, 2026) and then converted to euros using the average exchange rate for 2025 (ECB, 2026).

5.3.1 Climate impact

Reference values for the monetary valuation of climate impact can be interpreted as the costs of mitigating climate impact. The compiled reference values are presented in Table 9 together with the corresponding references. They are expressed as costs per tonne of CO_2e (tCO_2e) and range from 26 to 669 €/tCO_2e. The reference values reflect a range of valuation approaches, including social cost estimates, policy-based benchmarks, market prices, mitigation costs, and organisation-specific valuations. Most fall within the range of approximately 100-400 €/tCO_2e. Lower values are typically associated with market-based and mitigation-cost approaches and higher values represent more ambitious policy or strategic perspectives. Furthermore, the time horizon influences these figures. Mitigation-cost benchmarks often increase substantially over time (Trafikverket, 2024), reflecting increasing marginal cost as lower-cost mitigation options are implemented first. Thus, the ASEK 8.0 value for 2024 represents one point along an increasing cost trajectory towards 2045. Meanwhile, long-term social cost estimates take a global perspective and value the future damages caused by emissions today. These estimates depend partly on how heavily future impacts are weighted relative to present-day costs, with approaches that give greater weight to the welfare of future generations resulting in substantially higher reference values. Consequently, the choice of reference value should reflect both the purpose and time horizon of the assessment, as present-day values may not adequately represent the long-term value of reducing emissions.

The WSRs derived from historical MCA studies range from approximately 200 to 6000 €/tCO_2e (Table 10), considerably exceeding many of the compiled reference values. While some variation is expected due to differences in priorities and assessment contexts, this comparison suggests that climate impact has often been assigned valuations that are difficult to justify in relation to available reference values. One possible explanation is that climate impact is often given a high weight because of its perceived importance, while the actual difference in climate performance between alternatives is small. This combination can result in disproportionately high WSRs. These findings illustrate the potential of WSRs and reference values to support more transparent, consistent, and well-grounded weighting decisions in MCA.

Table 9. Reference values for climate impact, inflation-adjusted to 2025 price levels and converted to EUR.

Title	Reference value (€/tCO₂e)	Year original value	Reference
Comprehensive evidence implies a higher social cost of CO2	194	2020	(Rennert et al., 2022)
ASEK 8.0 action cost approach (road/railway traffic for 2024)	156	2019	(Trafikverket, 2024)
ASEK 8.0 action cost approach (road/railway traffic for 2045)	569	2019	(Trafikverket, 2024)
ASEK 8.0 action cost approach (aviation and shipping for 2024)	107	2019	(Trafikverket, 2024)
ASEK 8.0 action cost approach (aviation and shipping for 2045)	432	2019	(Trafikverket, 2024)
Klimatklivet (Swedish Environmental Protection Agency)	35	2024	(Naturvårdsverket, 2024)
Carbon tax in the Netherlands	94	2023	(Statistics Netherlands (CBS), 2024)
Statistics on emissions allowance trading	66	2024	(Naturvårdsverket, 2025)
Carbon tax in Sweden	136	2025	(The Swedish Government, 2025)
Updated estimates of social cost of carbon, revised DICE model	239	2016	(Nordhaus, 2017)
Environmental Prices Handbook, CE Delft	133	2015	(Sander et al., 2018)
Mitigating climate change (International Monetary Fund) – 2030	26	2020	(Fund, 2020)
Mitigating climate change (International Monetary Fund) – 2050	100	2020	(Fund, 2020)
Gryaab's climate strategy, acceptance level for actions	273	2024	(Lorick and Videbris, 2024)
Gryaab's climate strategy, level requires further investigation of actions	637	2024	(Lorick and Videbris, 2024)
Economic valuation of socio-economic benefits (Gothenburg city, Sweden)	398	2025	(Loman, 2025)
Economic valuation of socio-economic benefits (Gothenburg city, Sweden)	669	2025	(Loman, 2025)

Table 10. WSRs derived from previous MCA studies, inflation-adjusted to 2025 price levels and converted to EUR.

	Weighted score ratios						
Title	CO_2e (€/tCO_2e)	Energy (€/kWh)	N (€/kgN)	P (€/kgP)	BOD (€/$kgBOD_5$)	Year orig. value	Reference
Resource recovery in the Baltic Sea region (Fyris. Sweden)	6 164		40	350		2020	(Johannesdottir et al., 2021)
Resource recovery in the Baltic Sea region (Slupia. Poland)	1 558		66	562		2020	(Johannesdottir et al., 2021)
Sustainability assessment. Process invest. for Nya Rya (Gothenburg. Sweden)	205	0.02		0.86	0.017[1]	2022	(Lorick et al., 2022)
Pre-study pharmaceutical removal at Rya WWTP (Gothenburg. Sweden)	1 421	0.42				2023	(Neth et al., 2023)
Wastewater management in Kalmar (Kalmar. Sweden)		0.99				2011	(Arnell et al., 2011)
System study for wastewater and bio-waste in the Gothenburg region (Sweden)	767	0.57				2007	(Andersson et al., 2007)
Leachate management. Rya WWTP (Gothenburg, Sweden)	282	0.52	453	11073		2015	(Malmqvist et al., 2015)

[1] The BOD_7 value (0.020 €/$kgBOD_7$) was converted to BOD_5 by dividing with 1.15 (Helcom, 2011).

5.3.2 Energy use

Reference values for the monetary valuation of energy use are expressed as costs per kWh and presented in Table 11. Since the criterion energy use concerns energy as a resource, the aim was to identify reference values that reflect the resource value and energy scarcity rather than short-term energy prices. Most of the compiled values are therefore based on the costs of expanding electricity generation capacity, expressed as the levelized cost of energy (LCOE), which represents the lifetime cost of electricity generation per unit of electricity produced. For comparison, one reference value based on an actual electricity price is also included, although it should not be interpreted as a valuation of energy as a resource. The compiled values range from 0.035 to 0.087 €/kWh, excluding the electricity-price reference value.

The WSRs for energy use derived from historical MCA studies range from 0.02 to 0.99 €/kWh (Table 10). While generally of the same order of magnitude as the compiled reference values, several WSRs exceed the LCOE-based values, indicating that energy use has often been valued high in previous assessments.

Table 11. Reference values for energy use, inflation-adjusted to 2025 price levels and converted to EUR.

Title	Reference value (€/kWh)	Year original value	Reference
Levelized Cost of Energy for nuclear power	0.062	2021	(Elmqvist, 2021)
Levelized Cost of Energy for onshore wind power	0.035	2021	(Elmqvist, 2021)
Levelized Cost of Energy for solar parks	0.047	2021	(Elmqvist, 2021)
Levelized Cost of Energy for solar installations	0.087	2021	(Elmqvist, 2021)
Levelized Cost of Energy for waterpower	0.057	2021	(Elmqvist, 2021)
The socio-economic potential of energy efficiency improvements in buildings	0.044	2020	(Anthesis et al., 2021)
Average electricity price at Rya WWTP 2025	0.11	2025	(Gryaab, 2025)

5.3.3 Nitrogen discharge

The reference values for the monetary valuation of nitrogen are expressed as costs per kg of nitrogen discharged to receiving waters and are presented in Table 12, ranging from 0.0011 to 55 €/kgN. The objective was to identify reference values that reflect the societal and environmental costs of nitrogen emissions rather than the operational costs for nitrogen removal. The reference values represent different valuation approaches, including estimates of the social cost of nitrogen associated with environmental damage in aquatic ecosystems, nitrogen taxes applied to WWTP effluents, mitigation costs required to meet targets of the Baltic Sea Action Plan (BSAP), but also estimated costs of reducing nitrogen discharges. Despite this wide range, most reference values are below 14 €/kgN, with many falling within the range of 3-9 €/kgN. The variation may reflect differences in methodology, type of valuation and geographical context.

The WSRs derived from historical MCA studies range from 40 to 453 €/kgN (Table 10) and are generally substantially higher than the compiled reference values. This illustrates that the implicit valuation of nitrogen in these studies was generally high relative to the compiled reference values.

Table 12. Reference values for nitrogen discharged to receiving waters, inflation-adjusted to 2025 price levels and converted to EUR.

Title	Reference value (€/kgN)	Year original value	Reference
Estimation of the social cost of nitrogen, lower estimate	0.0011	2016	(Keeler et al., 2016)
Estimation of the social cost of nitrogen, upper estimate	11	2016	(Keeler et al., 2016)
Costs and benefits of nitrogen in the environment, lower/upper estimate	13/40	2011	(Brink et al., 2011)
The Danish emission tax for wastewater	4.5	2025	(Skatteministeriet, 2025)
Willingness to Pay for Improved Groundwater in PEI, Canada	0.72	2021	(Withey et al., 2025)
Swedish Environmental Protection Agency policy instruments for increased treatment at WWTPs	14	2010	(Naturvårdsverket, 2012)
Examples of polluter pays taxes, charges and fees in place in the EU	1.9	2013	(European commission, 2021)
Environmental Prices Handbook, CE Delft	3.9	2018	(Sander et al., 2018)
Willingness to pay in Sweden to meet Baltic Sea Action Plan targets	8.8	2014	(Söderqvist et al., 2017)
Willingness to pay in the nine Baltic Sea countries to meet Baltic Sea Action Plan targets	55	2014	(Söderqvist et al., 2017)

5.3.4 Phosphorus discharge

The reference values for the monetary valuation of phosphorus discharges are expressed as costs per kg of phosphorus discharged to receiving waters and are presented in Table 13. The compiled values include estimates of the social cost of phosphorus, phosphorus taxes applied to WWTP effluents, mitigation costs, and estimates of the costs of reducing phosphorus discharges to inland waters to meet the targets of the BSAP. The values show substantial variation, ranging from 4.4 to 1250 €/kg P.

The variation may reflect differences in valuation approaches, geographical contexts, and policy objectives. While small phosphorus reductions can often be achieved at relatively low cost, more advanced removal typically requires substantial investments, resulting in higher marginal costs. Willingness-to-pay estimates also vary considerably between regions (Söderqvist et al., 2017), whereas damage-based estimates are generally lower because they reflect environmental impacts rather than the costs of achieving policy targets. Overall, the wide range of values suggests that no single monetary valuation of phosphorus discharges is universally applicable.

The WSRs for phosphorus also span a wide range, from 0.9 to 11 000 €/kg P (Table 10), reflecting substantial variation in the implicit valuation of phosphorus across the included MCA studies.

Table 13. Reference values for phosphorus discharged to receiving waters, inflation-adjusted to 2025 price levels and converted to EUR.

Title	Reference value (€/kgP)	Year original value	Reference
The Danish emission tax for wastewater	25	2025	(Skatteministeriet, 2025)
Swedish Environmental Protection Agency policy instruments for increased treatment at WWTPs	644	2010	(Naturvårdsverket, 2012)
Examples of polluter pays taxes, charges and fees in place in the EU	4.4	2013	(European commission, 2021)
Environmental Prices Handbook, CE Delft	5.9	2018	(Sander et al., 2018)
Willingness to pay to meet Baltic Sea Action Plan targets, value for Bottenviken	74	2014	(Söderqvist et al., 2017)
Willingness to pay to meet Baltic Sea Action Plan targets, value for Bottenhavet	1 250	2014	(Söderqvist et al., 2017)
Willingness to pay to meet Baltic Sea Action Plan targets, value for Norra Östersjön	584	2014	(Söderqvist et al., 2017)
Willingness to pay to meet Baltic Sea Action Plan targets, value for Södra Östersjön	374	2014	(Söderqvist et al., 2017)
Willingness to pay to meet Baltic Sea Action Plan targets, value for Västerhavet	259	2014	(Söderqvist et al., 2017)
Willingness to pay in Sweden to meet Baltic Sea Action Plan targets	52	2014	(Söderqvist et al., 2017)
Willingness to pay in the nine Baltic Sea countries to meet Baltic Sea Action Plan targets	325	2014	(Söderqvist et al., 2017)

5.3.5 BOD discharge

Only one reference value for the monetary valuation of BOD discharge was identified: the Danish emission tax on BOD discharges from WWTPs, set at 2.5 €/kg BOD_5 (Skatteministeriet, 2025). The scarcity of reference values may reflect the fact that BOD-related impacts often overlap with those of other pollutants, particularly nitrogen and phosphorus, making it difficult to attribute environmental impacts specifically to BOD. As a result, valuation studies tend to focus on nutrients the effects of eutrophication rather than organic matter loads. The single WSR identified for BOD was 0.017 €/kg BOD_5 (Table 10), considerably lower than the reference value, although the comparison is based on only one value of each.

5.4 Application of the sustainability assessment method

The proposed sustainability assessment method (Section 5.2) was applied to case studies that represent different WWM decision problems. The applications provided a basis for evaluating and further refining the method.

The following case studies are presented:

- **Case I**: Assessment of technologies for pharmaceutical residues removal at the Rya WWTP, applying the proposed method with iterative calculation of WSRs (Section 5.2.8.1) (Paper III).
- **Case II**: Assessments of technologies for pharmaceutical residues removal at the Rya WWTP and approaches to resource recovery in the Baltic Sea region, applying the monetary-based weighting procedure (Section 5.2.8.4) (Paper IV).
- **Case III**: Assessment of process configurations for the expansion of the Rya WWTP, applying the proposed method with iterative calculation of WSRs (Section 5.2.8.1) and evaluating the use of subgroups for both scoring and weighting.

5.4.1 Case study I: Pharmaceutical residues removal at the Rya WWTP

Case study I was conducted as a sustainability assessment within a pre-study evaluating three technologies for the removal of pharmaceutical residues at the Rya WWTP (described in Section 2.1.3). The technologies were: ozonation, powdered activated carbon (PAC), and granular activated carbon (GAC). The aim of the assessment was to identify the most and least sustainable alternatives and determine which criteria had the greatest influence on the total sustainability scores.

The following methodological elements of the proposed method were applied, with particular emphasis on evaluating the iterative calculation of WSRs: (i) a scoring scale anchored to future acceptable levels, (ii) scoring before weighting, (iii) linearity checks for quantifiable criteria, (iv) calculation of WSRs and comparison with reference values, (v) calculation of values per weighted score step, and (vi) sensitivity analyses that separately favour each alternative.

5.4.1.1 Assessment assumptions and conditions

This assessment was conducted by a moderator and a project group consisting of five employees of the WWTP with different areas of expertise (operative manager, R&D manager, manager of project and planning, project leader and process specialist). This group performed both scoring and weighting, acting as experts during scoring and as decision-makers during weighting. The moderator facilitated the process but did not participate in scoring or weighting.

The alternatives were specified as full-scale implementations integrated into the existing treatment process and all three technologies were assumed to provide the same removal efficiency of pharmaceutical residues. The assessment focused on the operational phase, although selected construction and installation impacts were also estimated and included to illustrate their contribution relative to the operational impacts. Baseline conditions are summarised in Table 14. Most costs were originally in Swedish kronor (SEK) but are presented here in EUR, assuming an exchange rate of 10 SEK = 1 €. Costs reported in US dollars (USD) were converted to EUR using an exchange rate of 1 USD = 1 €.

Table 14. Case I: Baseline conditions for the sustainability assessment of ozonation, PAC, and GAC, including variants considered in the sensitivity analyses.

	Ozonation	PAC	GAC
Removal efficiency	Same removal efficiency of pharmaceutical residues assumed (80-90 %)[1]		
Location	Existing WWTP site	Existing WWTP site	In adjacent Rya Forest
Construction and installation	Estimates of climate impacts and costs from construction and installation are included.		
Treated wastewater flow	A yearly average flow of 4.08 m^3/s (129 Mm^3/year)		
Electricity mixture	Swedish electricity mix[2]		
Activated carbon origin	No activated carbon used	Fossil-based[3]	Fossil-based[3]
Dosage	10 mgO_3/l	15 mg PAC/l	20 000 bed volumes[4]
Heat production	11 400 MWh/year from local oxygen production	Estimated to 5 400 MWh/year from incineration of PAC sludge	No heat production
Transports	All transports with trucks using diesel as fuel, unless otherwise specified.		
Specific conditions for each alternative	Local oxygen production	New separate handling of PAC sludge included	Carbon reactivation in Belgium[5]

[1] Potential effects on the removal of other substances are not included in the assessment.
[2] Sensitivity analysis using alternative electricity mixes.
[3] Sensitivity analysis using activated carbon of renewable origin.
[4] Sensitivity analysis assuming 30 000 bed volumes.
[5] Sensitivity analysis assuming activated carbon reactivation at a new reactivation plant at Rya WWTP.

5.4.1.2 Scoring scale, criteria set and data acquisition

The scoring scale from 1 to 5 presented in Table 8 was applied, with score 3 representing the future acceptable level, defined specifically for each criterion. The selected environmental, economic, and social sustainability criteria are presented in Table 15 together with short descriptions and units (for quantifiable criteria). Data from the pre-study and other project documentation, calculations, and reference facilities with similar treatment processes were used to support the scoring of criteria. Quantifiable data were calculated on an annual basis. For non-quantifiable criteria, such as work environment, technical function, and reliability, the pre-study was also informed by experience from full-scale reference facilities in Linköping (ozonation) and in Simrishamn (ozonation and GAC), both in Sweden.

Climate impact was estimated by summarising contributions from construction, transport, and operation. For activated carbon of fossil origin, reported climate impacts vary greatly in the literature (Joseph et al., 2020). A value of 13.16 kg CO_2e/kg carbon was used (CONTACTICA and EMIVASA, 2018).

5.4.1.3 Scoring and weighting

The scores assigned by the assessment group, together with a brief justification for each criterion, are presented in Table 15. For the quantifiable criteria, linearity plots were developed iteratively during the scoring process, with the final plots shown in Figure 9. For climate impact, score 1 corresponds to approximately 15 000 tCO_2e/year. The PAC alternative with a climate impact of 25 000 tCO_2e/year exceeded this value but was nevertheless assigned the minimum score of 1, resulting in non-linear scoring. A similar situation occurred for chemical use. For sludge to agriculture, two alternatives had the same annual amount of sludge applied to agriculture and therefore received the same score.

The group performed the weighting according to Section 5.2.8, using iterative calculation of WSRs supported by a few reference values, such as 700 €/tCO_2e (Trafikverket, 2020) for climate impact and 0.08 €/kWh (the total electricity price at the Rya WWTP in 2019) for energy use. The monetary-based weighting procedure described in Section 5.2.8.4 was not applied, and the complete set of reference values in Section 5.3 had not yet been compiled. The final criterion weights, including minimum and maximum values, considered reasonable by the group, are presented in Table 16. The table also includes the WSRs, calculated according to Eq. 2, (with annual cost as criterion A), reference values for climate impact and energy use, and values per weighted score step for all criteria (with annual cost as criterion C).

Table 15. Criteria for Case I, grouped by sustainability dimension, with descriptions, assigned scores, and scoring justifications. Scores range from 1 to 5, with higher scores indicating better sustainability performance.

Sustainability dimension	Criterion [unit]	Description	Score			Scoring justification
			Ozon	PAC	GAC	
Environmental	**1. Climate impact [tCO_2e/year]**	Greenhouse gas emissions.	5	1	2	Score 5 for ozonation because removal of pharmaceutical residues is achieved with almost no increased climate impact. The group decided not to use linear scoring[1]. Score 1 was set to ≥15 000 tonne CO_2e/yr and PAC obtained score 1, even though the climate impact was distinctly higher. Linear scoring[1] of GAC.
	2. Energy use [kWh/year]	Energy use, considering energy as a resource. Climate impact from energy is included in criterion 1. Heat production is included in criterion 3.	1	5	4	Score 1 for ozonation due to its high energy consumption. Linear scoring[1] of PAC and GAC, both entailing relatively low energy consumption.
	3. Heat production [kWh/year]	Produced heat that can replace other heat production. Avoided climate impact from heat production is included in criterion 1.	5	4	3	All increases in heat production were considered >3 (acceptable level) and linear scoring[1] was applied.
	4. Chemical use [tonne/year]	Chemical use, regarding chemicals as a resource. Climate impact from chemicals is included in criterion 1.	5	1	3	Reduced chemical consumption is a current company goal, but +5% increase (GAC) was considered a future acceptable level (score 3). Non-linear scoring[1] applied and score 1 set to ≥1000 tonne chemicals/yr and PAC received score 1.
	5. Sludge to agriculture [tonne/year]	Quantity of sludge available to use as fertiliser on farmland.	5	2	5	A decreased quantity of sludge possible to use as fertiliser was considered below future acceptable level (score <3).
Social	**6. Organisational and legal work**	Organisational changes or new legal work or issues.	4	4	1	Shift work is not needed for any alternative. The organisational work of increasing staff by 1-3 persons for all three was considered score >3. GAC received score 1 due to probable legal work necessary to build in the adjacent Rya Forest.
	7. Society's perception	Society's perception of the alternatives and land use.	2	2	2	Score <3 for GAC due to proposed building in adjacent Rya Forest. Score <3 for ozonation because of high energy consumption and potential risks[2] which could be unpopular. Low score for PAC due to high chemical consumption and waste that needs to be incinerated.

Sustainability dimension	Criterion [unit]	Description	Score			Scoring justification
			Ozon	PAC	GAC	
	8. Work environment	Impact on the daily work environment at the WWTP, including dirt, physical demands, noise, and increased transports. Risks are included in criterion 9.	2	1	3	Score <3 for ozonation because of restrictions for certain people[3]. Low score for PAC because of dirt, dust, and increased number of transports.
	9. Safety risks	New safety risks to personnel and company property.	2	2	5	Scores for ozonation and PAC below future acceptable level (3) because both entail new security risks[4]. Score 5 for GAC since no new security risks appear.
Economic	**10. Annual cost [€/year]**	Annual operating and investment costs.	3	1	2	The annual cost for ozonation was considered acceptable (score 3) for achieving removal of pharmaceutical residues. Linear scoring[1] of PAC and GAC.
	11. References	Reference facilities and how established the technology is.	4	1	3	Many reference facilities for ozonation. Fewer for GAC than for ozonation. Reference facilities for this configuration of PAC were unknown to the project.
	12. Technical function and reliability	How robust, reliable, and technically demanding the technology is.	3	1	4	Score 1 for PAC due to large uncertainties regarding the treatment of PAC sludge. GAC considered as a robust process therefore score 4. Ozonation score 3 because more systems than for GAC that need to work.
	13. Maintenance	Extent of maintenance need. The cost of maintenance is included in criterion 10.	2	1	3	Maintenance needs for GAC were considered acceptable, but higher for ozonation due to multiple systems. Uncertainty regarding the maintenance need for PAC, both for the water process and the new PAC sludge treatment.
	14. Flexibility	How flexible, reversible, convertible the technology is.	2	4	1	PAC was considered very flexible as it could be stopped with limited remaining investment costs. Low score for GAC due to large investments in construction. Some investments in construction for ozonation although lower than for GAC.

[1] Linear and non-linear scoring is explained in 5.2.7.1.
[2] Potential risks with ozonation regarding by-products and transformation products formed, whose ecotoxicological effects are unclear (Sundin et al., 2017).
[3] Restrictions regarding staff with pacemaker.
[4] Ozonation entails fire risk and is harmful to health. PAC entails fire and explosion risk.

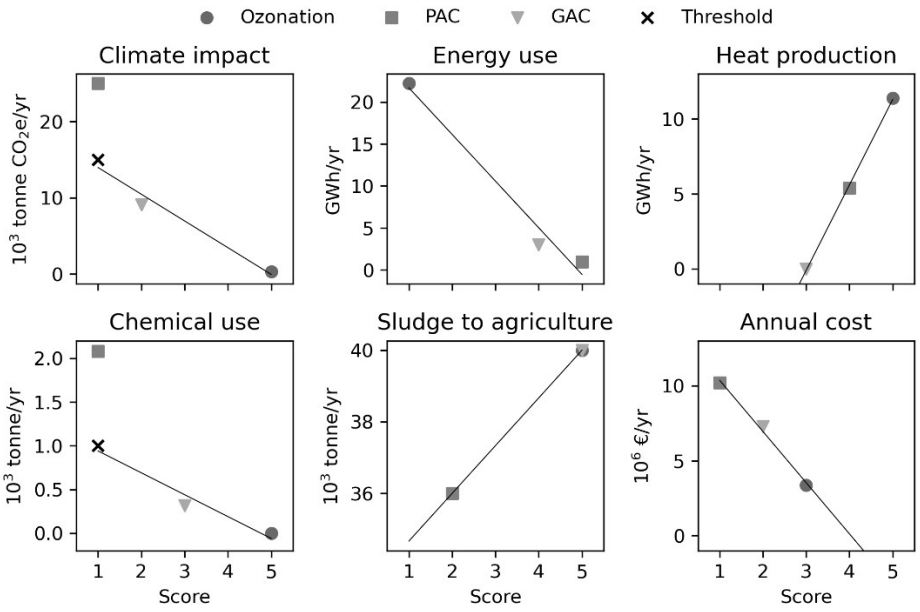


Figure 9. Case I: Linearity plots of performance data versus assigned scores for climate impact, energy use, heat production, chemical use, sludge to agriculture, and annual cost.

Table 16. Case I: Value per score step for quantifiable criteria, value per weighted score step for all criteria, WSRs relative to annual cost for quantifiable criteria, and criterion weights.

Criterion	Value per score step [unit/score step, yr]	Value per weighted score step [€/weighted score step, yr]	WSR [€/unit]	Normal-ised weight [%]
1. Climate impact	3515 tCO_2e	4.8 million	1 360 €/tCO_2e	18
2. Energy use	5 554 MWh	2.0 million	0.4 €/kWh	8
3. Heat production	5 715 MWh	0.7 million	0.1 €/kWh	3
4. Chemical use	250 t chemicals	0.7 million	2 730 €/t chemicals	3
5. Sludge to agriculture	1 333 t sludge	1.4 million	1 020 €/t sludge	5
6. Organisational and legal work	-	1.4 million	-	5
7. Society's perception	-	0.7 million	-	3
8. Work environment	-	2.7 million	-	11
9. Safety risks	-	2.7 million	-	11
10. Annual cost	3.4 million €	3.4 million	1	13
11. References	-	1.4 million	-	5
12. Technical function, reliability	-	2.0 million	-	8
13. Maintenance	-	0.7 million	-	3
14. Flexibility	-	1.4 million	-	5

5.4.1.4 Analysis of the results

Figure 10 presents the total sustainability scores for the alternatives, calculated according to Eq. 6, using the scores in Table 15 and the weights in Table 16. The height of each bar represents the total sustainability score, while the individual criterion contributions are shown as separate blocks. Larger blocks indicate criteria with a greater contribution to the total score, whereas

smaller blocks indicate a smaller contribution due to either a low criterion weight, a low score for that alternative, or both. The results for the original alternatives and all sensitivity analyses are shown in Figure 11. The sensitivity analyses included changes to the baseline conditions presented in Table 14 and the resulting changes in individual scores. These included the use of renewable instead of fossil-based activated carbon, a European instead of a Swedish electricity mix, increased GAC bed volumes, and the relocation of the GAC facility from Rya forest to within the WWTP site. In addition, combined weighting sensitivity analyses were performed to favour each alternative in turn, referred to as "best for alternative X" scenarios (Section 5.2.9.1).

Ozonation and GAC achieved similar total sustainability scores (3.2 and 2.9, respectively), with ozonation ranking slightly higher, while PAC received a substantially lower score (1.9) and was ranked as the least sustainable alternative. Even under weighting assumptions favouring PAC ("best for PAC"), it did not reach the same sustainability level as ozonation or GAC (see Figure 11). This was mainly because of its continuous consumption of activated carbon without regeneration, resulting in higher climate impact and chemical use, poorer work environment conditions, and sludge that cannot be applied to agricultural land. However, the assessed PAC configuration, involving dosing to an MBBR process, is novel in full-scale application, while previous studies of other PAC configurations have found PAC-based treatment to be competitive with ozonation under certain conditions. For instance, Margot et al. (2013) identified PAC followed by ultrafiltration as the most suitable option when receiving waters were particularly sensitive.

The close ranking of ozonation and GAC indicates that the preferred alternative depends on the relative importance assigned to individual sustainability criteria. Ozonation is favoured when climate impact is prioritised, whereas GAC performs better when energy use and technical performance are considered more important, consistent with previous LCA studies (Pesqueira et al., 2020). The resulting WSR for climate impact was relatively high, which is further discussed in Section 5.4.1.5. It should be noted that the assessment did not consider differences in pharmaceutical removal efficiency, additional removal of other pollutants, uncertainties related to maintenance requirements, PAC sludge treatment, or the potential formation of by- and transformation products from ozonation.

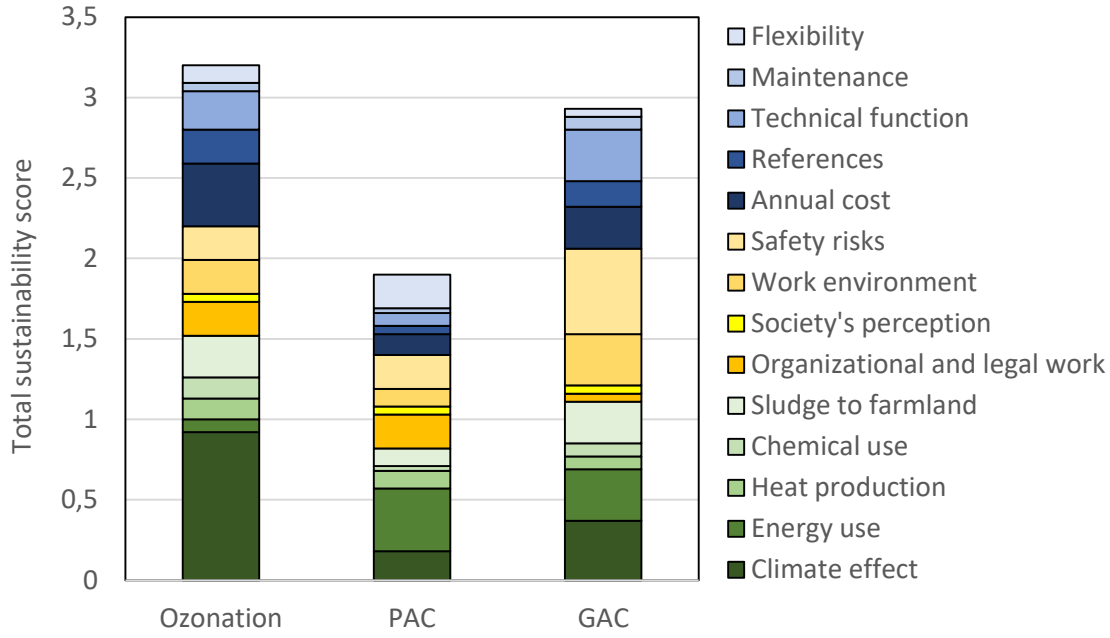


Figure 10. Case I: Total sustainability scores for ozonation, PAC and GAC. The height of each bar represents the total sustainability score, while each block shows the contribution of an individual criterion.

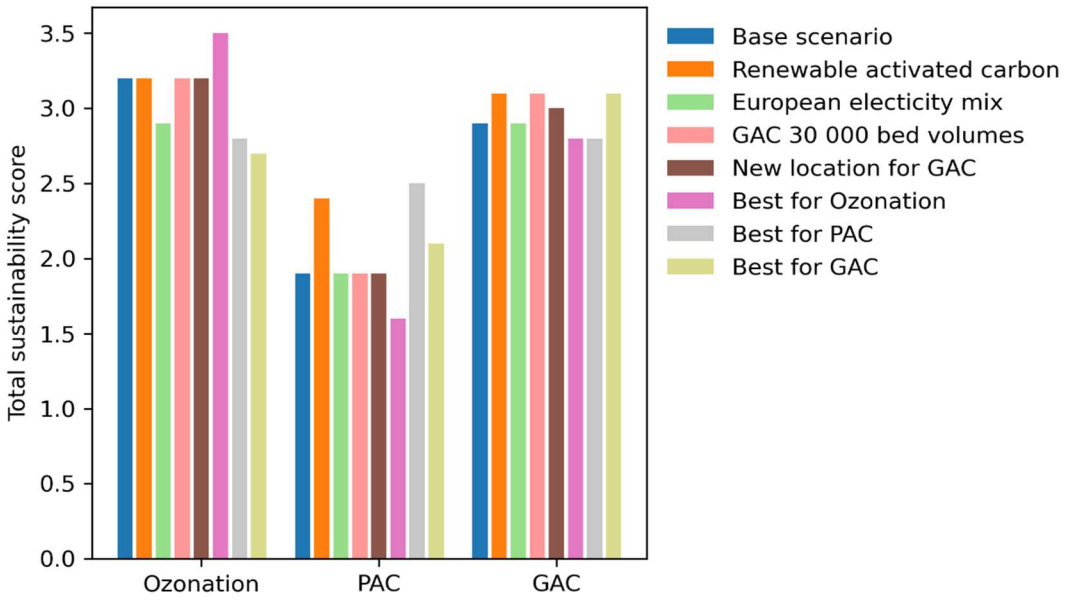


Figure 11. Case I: Total sustainability scores for ozonation, PAC and GAC in the base case and all sensitivity analyses.

5.4.1.5 Reflections on the sustainability assessment method with WSRs

Using a scoring scale related to future acceptable levels worked well in practice. Discussions mostly began by defining what should be regarded as an acceptable future performance level (score 3) after which at least one additional point on the scale was defined, creating common reference points for scoring the alternatives. Performing the scoring before the weighting was necessary for calculating WSRs and proved practical in the assessment process. The scores, their underlying data and the reasoning behind them were available during the subsequent weighting process, making it possible to understand what had been considered for each criterion. The linearity checks were valuable for quantifiable criteria, helping to ensure that score differences reflected proportional differences in the underlying data. However, climate impact and chemical use were scored non-linearly. Rather than extending the 1–5 scale to accommodate an alternative with substantially different performance, the group chose to assign it the minimum score of 1, even though its performance fell outside the range represented by the scale. This illustrates the need for some flexibility when applying the scale. In this case, the approach favoured PAC, which would otherwise have received an even lower score.

Scoring non-quantifiable criteria was generally more challenging, as several aspects often had to be balanced within a single criterion. Requiring participants to justify each assigned score proved valuable, as it made the reasoning transparent and facilitated subsequent discussions. For quantifiable criteria, calculating the value corresponding to one score step also supported the scoring process by allowing the group to assess whether the difference in performance represented by one score step was reasonable. A similar approach was used during weighting for non-quantifiable criteria through the value per weighted score step. For example, when weighting work environment, the group could consider whether an improvement of one score step was reasonably valued at the corresponding increase in annual cost. Although participants initially found this measure less intuitive, it provided a concrete basis for discussing the weighting of non-quantifiable criteria. The introduction of WSRs for quantifiable criteria made the implications of the assigned criterion weights visible to the assessment group, providing an additional perspective for reflecting on them. The group generally preferred to assign weights directly based on their priorities and made only minor adjustments based on the calculated WSRs.

The sensitivity analyses further strengthened the assessment by demonstrating the robustness of the results under varying conditions. As noted by Belton and Stewart (2002), simple additive value models, such as the one applied here, combined with sensitivity analysis can often provide essentially the same results and insights as more complex MAUT models (referred to in Section 2.3.2). The combined weighting sensitivity analyses ("best for alternative X"), in which each alternative was separately favoured within predefined weight ranges, were particularly useful for examining how different priorities affected the ranking. Sensitivity analyses also made it possible to explore different priorities among the participants and to examine the effects of changes in the underlying data, scoring, and weighting, providing a flexible basis for discussion and decision-making. Overall, participants appreciated that the method produced considerably more information than a simple ranking of alternatives. By presenting scores and their underlying reasoning, criterion weights, WSRs, and sensitivity analysis results together, the assessment became more transparent and easier to interpret for those using the results. The possibility of revising input data or priorities and immediately exploring the consequences was also considered a valuable feature, making the method suitable not only for identifying the most sustainable alternative but also for supporting stakeholder dialogue and informed decision-making.

5.4.2 Case study II: Monetary-based weighting applied to two cases

Case study II comprised two cases and involved three assessment groups. Case A was based on the study of pharmaceutical residues removal at the Rya WWTP, also used in Case study I (Section 5.4.1), whereas Case B concerned resource recovery in the Baltic Sea region. The aim was to demonstrate and evaluate the proposed monetary-based weighting procedure (Section 5.2.8.4) by comparing it with a conventional weighting approach using three assessment groups. This enabled a comparison of the resulting criterion weights for each case both between the two weighting methods and among the assessment groups.

5.4.2.1 Description of groups, cases and weighting procedures

Three workshops, one for each assessment group, were led by two moderators. The groups were:

- **WWTP group (W):** Six employees at Gryaab, the company operating Rya WWTP (Section 2.1.3). The participants represented a range of engineering professions.
- **University group (U):** Twelve researchers from Chalmers University of Technology, divided into two groups of six participants each (U1 and U2). The participants were PhD students, research assistants, postdoctoral researchers, and professors.
- **Consultancy group (C):** Five employees of the same consultancy firm in Gothenburg, all with university-level education and professional experience in the water sector.

The cases were real assessments, chosen to reflect different objectives, authorship and criteria:

- **Case A:** The pre-study on pharmaceutical residues removal at the Rya WWTP, described in Section 5.4.1. The criteria considered were climate impact and energy use.
- **Case B:** A case study on resource recovery in the Baltic Sea region (Fyrisån river catchment, Uppsala, Sweden) based on an MCA study by Johannesdottir et al. (2021). The evaluated alternatives were labelled baseline (current WWM), incineration, nutrient extraction, and source separation. The criteria considered were climate impact and eutrophication potential, with the latter disaggregated into nitrogen and phosphorus discharges to receiving waters when using the monetary-based weighting procedure.

All assessment groups received a brief presentation of the two cases, including the alternatives considered and the MCA methodology. They were then assigned two weighting tasks. First, the groups applied a conventional weighting procedure by assigning weights to the selected criteria while keeping the weights of the remaining criteria from the original assessments fixed. The weighting scale had no predefined upper limit, although the existing criterion weights served as reference values. Second, the groups applied the monetary-based weighting procedure (Section 5.2.8.4) by determining monetary valuations for climate impact, energy use, and nitrogen and phosphorus discharges to receiving waters. To support the valuation process, they were provided with the compiled monetary reference values presented in Section 5.3.

5.4.2.2 Conventional and monetary-based weights for Case A

The WWTP group began by conventionally weighting energy use in Case A, emphasising that high energy consumption increases system vulnerability, may require backup power and grid expansion, and conflicts with organisational energy-efficiency goals. Despite initial disagreement over the appropriate weight, the group agreed on a weight of 5. For the monetary valuation, the group selected 0.023 €/kWh, keeping it below the estimated cost of expanding energy production (Elmqvist, 2021) while accounting for increased system vulnerability. Climate impact was unanimously assigned very high importance and given a conventional weight of 20. For the monetary valuation, the group considered the Nordhaus reference value (Nordhaus, 2017) and the WSR from a previous project (Lorick et al., 2022) as the main benchmarks. Although some participants proposed the slightly higher value in the organisation's climate strategy (Lorick and Videbris, 2024), the group reached a compromise of 248 €/tCO_2e.

The university groups differed in their conventional weighting. In U1, climate impact received a weight of 25, reflecting its increasing societal and political relevance, while energy use was considered less critical and assigned a weight of 5. In U2, climate impact received a weight of 17, reflecting its lower priority relative to cost, safety, and work environment, whereas energy use received a weight of 23 because of anticipated future constraints in energy distribution. The monetary valuations of climate impact also differed between U1 and U2. In U1, the discussion focused on whether to adopt Nordhaus's reference value (Nordhaus, 2017) or the much lower emissions trading allowance price (Naturvårdsverket, 2025). While some supported the emissions trading allowance price as a market benchmark, most argued that municipalities with sufficient resources should prioritise direct mitigation, and U1 agreed on 226 €/tCO_2e. U2 selected 316 €/tCO_2e, remaining below the lower reference value reported by the City of Gothenburg (Loman, 2025). For energy use, U1 adopted 0.18 €/kWh, in the same range as the case study organisation's electricity price (0.11 €/kWh), while U2 selected 0.13 €/kWh, close to the same reference value, although the group had assigned energy use a high conventional weight.

The consultancy group assigned climate impact the highest conventional weight (30), reflecting its importance in relation to political and societal goals and giving it a higher weight than annual cost (25). Energy use received a moderate weight of 10, partly motivated by the fact that its climate impact was considered separately. For the monetary valuation, the consultancy group adopted 407 €/tCO_2e, close to the City of Gothenburg's lower reference value (Loman, 2025). The group argued that Sweden's economic capacity and environmental commitments justified a value above the Swedish carbon tax (The Swedish Government, 2025), the Nordhaus reference value (Nordhaus, 2017), and the lower value in the organisation's climate strategy (Lorick and Videbris, 2024). Energy use was valued at 0.063 €/kWh, reflecting its value as a limited resource beyond the electricity price.

Table 17 summarizes the weights for climate impact and energy use assigned by the groups, including the conventionally set weights, their corresponding normalised weights and calculated WSRs, followed by the monetary valuations (€/tCO_2e or €/kWh) and the resulting normalised weights from the monetary-based weighting approach. The weights and WSRs from the original study are included for comparison. Normalised weights were calculated according to Eq. 5, using the weights from the original study for the remaining criteria. Coefficients of variation (CVs) are reported for the WSRs and monetary valuations of each criterion across the assessment groups. Figure 12 presents the resulting total sustainability scores, calculated according to Eq. 6 for each alternative and assessment group (W, U1, U2 and C) under both weighting procedures.

Table 17. Case A: Assigned and normalised weights and WSRs from conventional weighting, and monetary valuations and resulting normalised weights from monetary-based weighting, for climate impact and energy use across the four groups (W, U1, U2 and C). Coefficients of variation (CVs) for WSRs and monetary valuations are reported across the groups. Weights and WSRs from the original study are included for comparison.

		Conventional weighting			**Monetary-based weighting**	
Criterion	**Group**	**Weight**	**Normalised weight[1] (%)**	**WSR (€/tCO_2e or kWh)**	**Monetary valuation (€/tCO_2e or kWh)**	**Normalised weight[1] (%)**
Climate impact	Original study	35	18	1421	-	-
	W	20	12	812	248	4
	U1	25	15	1015	226	4
	U2	17	9	690	316	5
	C	30	17	1218	407	7
	CV (W, U1, U2, C):			25%	27%	
Energy use	Original study	15	8	0.39	-	-
	W	5	3	0.13	0.023	1
	U1	5	3	0.13	0.18	5
	U2	23	13	0.59	0.13	3
	C	10	6	0.26	0.063	2
	CV (W, U1, U2, C):			78%	70%	

[1] Normalised weights are calculated across all included criteria, not only those shown here; therefore, the weights presented in this table do not sum to 100%.

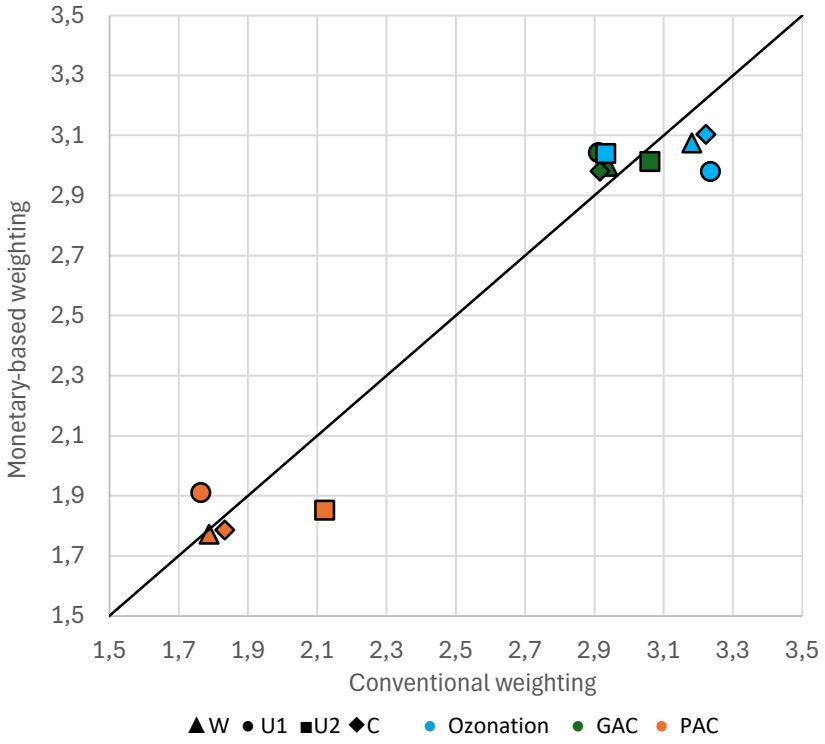


Figure 12. Comparison of total sustainability scores for Case A obtained using the monetary-based and the conventional weighting method. The black line represents the line of equality. Colours represent the alternatives (PAC, GAC, and ozonation), while marker shapes (triangle, circle, square, and diamond) represent the assessment groups.

5.4.2.3 Conventional and monetary-based weights for Case B

The WWTP group found it challenging to conventionally weight eutrophication potential in Case B, partly because the criterion was not disaggregated into nitrogen and phosphorus and because information on recipient sensitivity and regulatory requirements was limited. Nevertheless, the group agreed on a weight of 10. As in Case A, climate impact was considered an important criterion and assigned high priority. Although proposed weights ranged from 10 to 15, the group agreed on a conventional weight of 12 relative to the other criteria. For the monetary valuations, the group retained the climate impact valuation from Case A (248 €/tCO_2e). For nitrogen and phosphorus discharges, the Swedish Environmental Protection Agency's mitigation cost estimates were considered the most relevant reference values (Naturvårdsverket, 2012), resulting in valuations of 14 €/kgN and 640 €/kgP.

The university groups, U1 and U2, assigned identical conventional weights to climate impact (13) and eutrophication potential (19), reflecting the shared view that, given the sensitivity of the recipient, eutrophication should be prioritised over climate impact. Both groups also retained their climate impact valuations from Case A, resulting in values of 226 €/tCO_2e for U1 and 316 €/tCO_2e for U2. The monetary valuations of nitrogen and phosphorus discharges differed between the two university groups. U1 selected a relatively high phosphorus valuation of 904 €/kgP, informed by the importance of phosphorus for the recipient and the reference value based on willingness to pay for achieving good water status in the Bottenhavet water district (Söderqvist et al., 2017). Nitrogen was valued at 14 €/kgN, in line with the Swedish Environmental Protection Agency's mitigation cost estimates (Naturvårdsverket, 2012). In contrast, U2 adopted a lower phosphorus valuation of 407 €/kgP, without a clear reference value as a basis, and 4.8 €/kgN, based on social cost estimates and emission taxes.

The consultancy group emphasised the sensitivity of the recipient. Following discussion of the criterion and its relation to regulatory requirements, eutrophication potential was assigned a conventional weight of 10, equal to the weight of nutrient recovery, while climate impact received a weight of 15 after consideration of an even higher value. For the monetary valuations, the group retained the climate impact valuation from Case A (407 €/tCO_2e). Phosphorus was valued at 407 €/kgP, based on the Swedish Environmental Protection Agency's mitigation cost estimates (Naturvårdsverket, 2012) and regional willingness to pay for achieving good water status (Söderqvist et al., 2017), whereas nitrogen was valued at 14 €/kgN, partly based on the social cost estimates (Keeler et al., 2016).

Table 18 presents the results of both weighting procedures for climate impact and eutrophication potential across the four assessment groups, including weights and WSRs from the original study for comparison. Coefficients of variation (CVs) are reported for the WSRs and monetary valuations across the groups. Figure 13 presents the resulting total sustainability scores for all alternatives under both weighting procedures, calculated according to Eq. 6.

Table 18. Case B: Assigned and normalised weights and WSRs from conventional weighting, and monetary valuations and resulting normalised weights from monetary-based weighting, across the four assessment groups (W, U1, U2 and C). Coefficients of variation (CVs) for WSRs and monetary valuations are reported across the groups. Weights and WSRs from the original study are included for comparison.

		Conventional weighting			Monetary-based weighting	
Criterion	**Group**	**Weight**	**Normalised weight[1] (%)**	**WSR (€/tCO_2e or kgN or kgP)**	**Monetary valuation (€/tCO_2e, kgN or kgP)**	**Normalised weight[1] (%)**
Climate impact	Original study	-	10	6164	-	-
	W	12	13	8882	248	1
	U1	13	13	9622	226	1
	U2	13	13	9622	316	1
	C	15	16	11102	407	1
		CV (W, U1, U2, C):		9%	27%	
Nitrogen discharge to water	Original study	-	4	40	-	-
	W	8	9	89	14	2
	U1	15	15	169	14	2
	U2	15	15	169	4.8	1
	C	8	9	89	14	2
		CV (W, U1, U2, C):		36%	39%	
Phosphorus discharge to water	Original study	-	1	355	-	-
	W	2	2	791	640	3
	U1	4	4	1503	904	4
	U2	4	4	1503	407	2
	C	2	2	791	407	2
		CV (W, U1, U2, C):		36%	40%	

[1] Normalised weights are calculated across all included criteria, not only those shown here; therefore, the weights presented in this table do not sum to 100%.

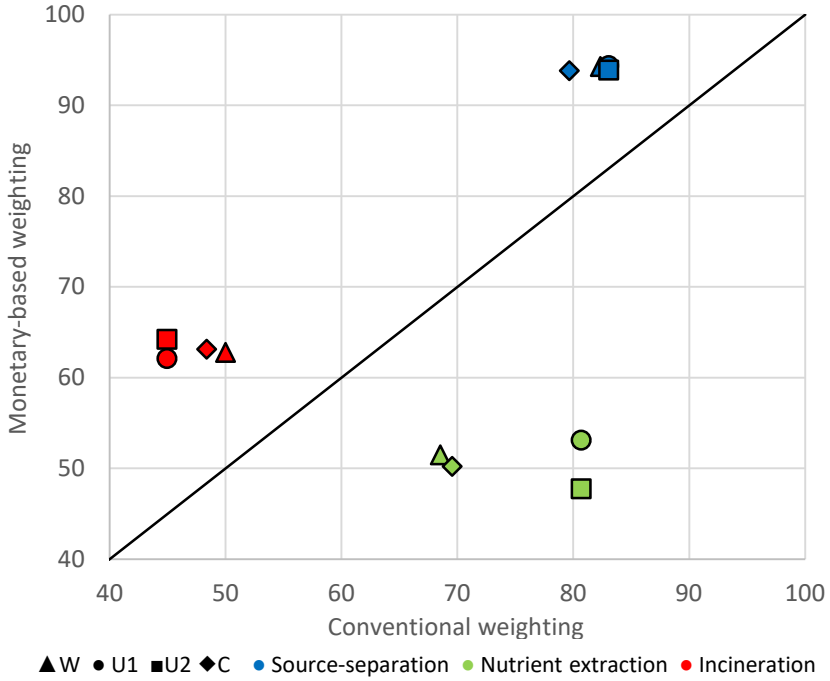


Figure 13. Comparison of total sustainability scores for Case B using the monetary-based and the conventional weighting method. The black line represents the line of equality. Colours represent the alternatives while marker shapes (triangle, circle, square, and diamond) represent the assessment groups.

5.4.2.4 Reflections on the application of the monetary-based weighting procedure

As the aim of Case Study II was to evaluate the monetary-based weighting procedure, the focus was on its application rather than on the assessment results. The monetary valuations varied across the assessment groups; however, for most of the criteria included in the monetary-based weighting, the resulting normalised weights were clearly lower than those obtained using conventional weighting (Table 17 and Table 18). Consequently, criteria that were included in the assessment but not assigned monetary-based weights gained greater relative influence on the total sustainability scores. This demonstrates the value of applying a consistent weighting procedure across all criteria in an assessment.

The variation in monetary valuations between the assessment groups was not unexpected, as the groups may have relied on different external reference values when assigning monetary values. The workshop discussions showed that differences between groups could be related to differences in perspectives, experience, and organisational context. For climate impact, conventional weighting often resulted in higher implicit valuations (WSRs) than those obtained through monetary-based weighting (Table 17 and Table 18), and relatively high WSRs were also derived from previous MCA studies (Table 10). A similar pattern was observed for nitrogen discharge and, to a lesser extent, for energy use and phosphorus discharge. While conventional MCA weighting also implies monetary valuations through the combination of weights and scores, these valuations remain implicit. The proposed monetary-based approach makes these valuations explicit and therefore easier to evaluate and discuss.

The total sustainability scores for both cases (Figure 12 and Figure 13) show that the weighting procedure can affect the ranking of alternatives. In Case B, the ranking changed for all assessment groups when monetary-based weighting was applied. In Case A, the ranking changed for U1 and U2, but the two leading alternatives had similar scores and the lowest-ranked remained unchanged.

However, assigning monetary values was not necessarily straightforward. Participants had to interpret, choose, and agree on values from a range of available references, particularly for criteria such as climate impact. These reference values reflected different contexts and assumptions, providing a basis for discussing and justifying the assigned valuations without removing the need for judgement. This suggests that the main value of the monetary-based procedure is not to identify an objectively correct valuation on which to base the weighting, but to make the basis for weighting more transparent and open to discussion.

5.4.3 Case study III: Capacity expansion at the Rya WWTP

Case III comprised two sustainability assessments conducted as part of a major expansion project of the Rya WWTP (Section 2.1.3), called the Nya Rya project. The assessments involved members of the Nya Rya project group as well as other employees at Gryaab and were facilitated by a moderator. The project involves the construction of new treatment facilities on land adjacent to the existing plant. The work described in Chapter 3 contributed to identifying the need for expansion and defining potential future treatment requirements. Alternative process designs were developed to meet demands regarding treatment capacity, influent loads, effluent quality, and available space. The assessments aimed to systematically compare the different process configurations and provide a common basis for decision-making. The selection of criteria was guided by the aim of providing a holistic sustainability assessment. The first assessment compared four process alternatives: activated sludge (AS), moving bed biofilm reactor (MBBR), aerobic granular sludge (AGS), and membrane bioreactor (MBR).

The two sustainability assessments were conducted in 2021 and 2022, both following the method described in Section 5.2, with WSRs calculated iteratively throughout the assessment process and used to different extents. In the first assessment, the scoring group, consisting of staff with different areas of expertise at Gryaab, was divided into two subgroups (S1 and S2), which independently scored the alternatives. The project group decided to use the mean of the scores assigned by the two subgroups (SM) for each criterion and alternative to calculate the baseline total sustainability scores. Another group, comprising participants with decision-making roles at Gryaab, was responsible for assigning weights to the criteria. This group was also divided into two subgroups (W1 and W2), which performed the weighting independently. Although both subgroups were introduced to the same weighting procedure, they adopted somewhat different approaches. W1 aimed to assign approximately one-third of the total weight to recipient impact, one-third to annual cost, and the remaining third to the other criteria. W2 instead considered each criterion independently, making greater use of the calculated WSRs. The original intention was for the two subgroups to subsequently assign weights jointly as one group. However, following a discussion, they decided to use the weights assigned by W1 to calculate the baseline total sustainability scores.

In the present work, to examine the influence of differences in scoring and weighting between the subgroups, total sustainability scores were calculated for different combinations of the subgroups' results, including the mean of the weights assigned by W1 and W2 (WM). The total sustainability scores for seven scoring and weighting combinations are presented in Table 19 and the WSRs calculated from the weights assigned by each weighting subgroup are presented in Table 20. Based on the overall results of this first sustainability assessment, together with other relevant considerations, the decision-makers chose to exclude the MBBR alternative.

Table 19. Total sustainability scores for the four process alternatives under combinations of the two weighting (W1 and W2) and scoring (S1 and S2) subgroups in the first Nya Rya sustainability assessment. SM and WM represent the mean scores and weights, respectively.

	Total sustainability scores						
	S1, W1	**S2, W1**	**SM, W1 (baseline)**	**S1, W2**	**S2, W2**	**SM, W2**	**SM, WM**
AS	3.5	3.5	3.5	3.4	3.5	3.5	3.5
MBBR	2.2	2.4	2.3	2.5	2.9	2.7	2.5
AGS	3.7	3.7	3.7	3.2	3.2	3.2	3.4
MBR	2.5	2.6	2.6	2.6	2.7	2.7	2.6

Before the second sustainability assessment, the remaining alternatives (AS, AGS and MBR) were further refined, and two combined-process alternatives (AS–MBR and AGS–MBR) were added. This resulted in a total of five alternatives, all designed to meet the requirements specified at this stage of the project. More detailed data had also been compiled for the second assessment. The participants differed from those in the first assessment, and no subgroups were used. The scoring group still comprised participants with different areas of expertise at Gryaab, while the weighting group mainly comprised participants with decision-making roles. The selection of criteria was slightly adjusted to better account for differences between the alternatives.

The baseline total sustainability scores from the second assessment are presented in Figure 14, while the corresponding WSRs, together with those from the first assessment, are presented in Table 20. Several sensitivity analyses were conducted to investigate how changes in conditions and priorities affected the results and to assess their robustness, but these analyses are not presented here. AGS received the highest total sustainability score and MBR the lowest, although the differences between the alternatives were relatively small.

Costs and valuations were originally in Swedish kronor (SEK) but are presented here in EUR, assuming an exchange rate of 10 SEK = 1 €.

Table 20. WSRs for the first and second Nya Rya sustainability assessments. Dashes (–) indicate criteria for which WSRs could not be calculated in the first assessment because the alternatives did not differ in their underlying data.

	1st sustainability assessment		**2nd sustainability assessment**
Criterion	**WSR, W1 [€/unit]**	**WSR, W2 [€/unit]**	**WSR [€/unit]**
Climate impact [tCO_2e]	147	588	202
Recipient impact, P [kgP]	-	-	0.86
Recipient impact, BOD [kgBOD]	-	-	0.02
Energy use [kWh]	0.02	0.18	0.02
Chemical use [t]	257	1027	387

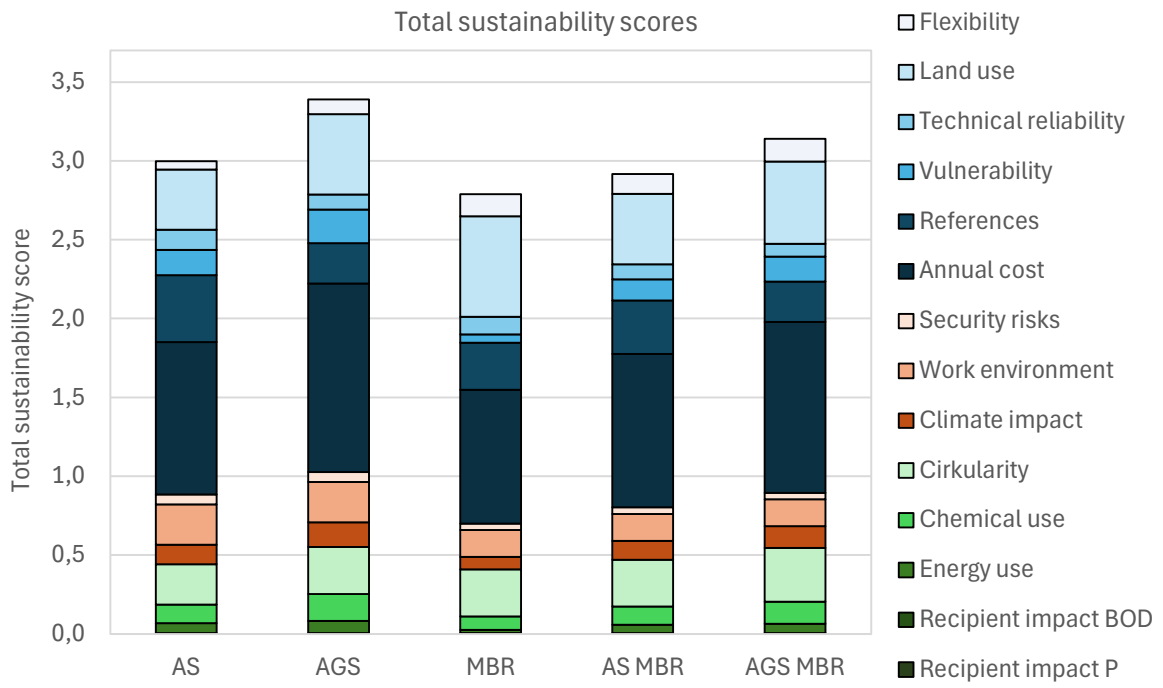


Figure 14. Baseline total sustainability scores for the five process configurations (AS, AGS, MBR, AS–MBR and AGS–MBR) in the second Nya Rya sustainability assessment. Coloured blocks show the contribution of each criterion.

5.4.3.1 Reflections on the sustainability assessment method with WSRs

The sustainability assessments in the Nya Rya project provided decision support as the design and available knowledge evolved. The assessments illustrated how assessment results can be influenced by scoring, weighting and changes in the underlying data and assumptions. In the first assessment, differences between the weighting subgroups affected the ranking of alternatives (Table 19), while the second assessment was based on refined process alternatives, more detailed data and a revised selection of criteria. The assessments provided a basis for comparing alternatives, identifying important trade-offs and discussing uncertainties, rather than treating the exact sustainability scores or rankings as definitive outcomes.

The first assessment demonstrated the influence of weighting, as the two weighting subgroups produced different results. Dividing participants into subgroups allowed different perspectives to develop independently and may also reduce the influence of particularly dominant individuals, allowing other views to emerge. However, the case also showed that how the results from different subgroups are subsequently selected or combined can affect the outcome and should therefore be transparent. In the first assessment, one participant had a strong influence on the final choice of weights. This highlights the potential value of repeating assessments with different group compositions or using subgroups.

The WSRs provided an additional perspective by making explicit the relative valuations implied by the assigned weights. The two weighting subgroups differed in their use of WSRs, with W2 making greater use of them when assigning weights. The weights from W1 were ultimately selected for the baseline assessment and their resulting WSRs were generally closer to those of the second assessment than the WSRs from W2 (Table 20). Examining the WSRs helped assess whether the implied trade-offs were reasonable. In the second assessment, WSRs were used more actively and iteratively. Towards the end of the weighting process, the participants revisited all criteria and revised some weights in light of the calculated WSRs. However, the WSR for chemical use was difficult to interpret due to the criterion's complexity and the lack of suitable reference values.

5.5 Summary and main findings

A sustainability assessment method for WWM was developed based on MCA and applied and refined through case studies. The method includes a scoring scale linked to future acceptable levels, linearity checks for quantifiable criteria, calculation of values per score step and WSRs, and structured sensitivity analyses. A monetary-based weighting procedure, supported by a compilation of monetary reference values, was also developed.

The case studies showed that the method provides a sustainability ranking of the included alternatives, insight into which criteria have the greatest influence on the results, and an understanding of how assumptions, priorities, trade-offs, and uncertainties affect the outcomes. Sensitivity analyses provided information on the robustness of the results, while WSRs revealed the implicit monetary valuations resulting from conventional weighting. Comparison with reference values showed that these implicit valuations can be relatively high. In monetary-based weighting, monetary valuations are directly assigned, allowing comparison with external reference values and providing a clearer basis for discussing and justifying weights and trade-offs. In Case Study II, differences between assessment groups remained, but the workshops showed that monetary-based weighting helped participants articulate and discuss their valuations. The procedure therefore does not eliminate subjective judgement but provides a more transparent basis for weighting.

Overall, the proposed method provides a structured and flexible basis for sustainability assessment across different WWM decision problems, allowing criteria and priorities to be adapted to the specific context. By using the proposed definition of sustainable WWM to guide criteria selection, the method supports holistic sustainability assessment and connects the definition to the evaluation of alternative WWM pathways.

6 Discussion

Identifying pathways to future sustainable WWM requires more than evaluating the environmental and economic performance of wastewater technologies, systems or strategies. It can be understood as involving three closely connected questions: What future should WWM be prepared for? What constitutes sustainable WWM? How can the sustainability of different pathways be assessed? An important contribution of this thesis lies not only in addressing each of these three questions, but also in showing how they complement each other and can be combined into a framework.

Preparing WWM for the future required exploring what that future may involve and what may become important or challenging in the decades ahead, making scenario analysis a natural starting point. Many future scenario studies use predefined or broad scenarios to support decision-making and strategy development, rather than focusing first on developing and defining the actual scenarios as precisely as possible. The analysis in this thesis began by identifying conditions that may be important for future WWM and then exploring how these conditions might develop over time. This approach was chosen to reduce the risk of overlooking conditions that could significantly affect future WWM. Based on the identified conditions and their possible developments, a range of plausible future scenarios was developed rather than only a single most likely one. This was important because, even if one scenario appears more likely than others, there is no guarantee that it will occur, and a less likely future may present considerably different challenges for WWM. Considering a range of plausible futures therefore shifts the focus from identifying solutions optimal for one expected future towards solutions that can remain sustainable under different and changing conditions. This is particularly relevant for WWM, where infrastructure investments have long lifetimes, during which wastewater flows and loads, regulatory requirements, climate conditions, resource availability, and societal priorities may change. Quantifying future scenarios allows them to be used in process models, linking possible future developments to their implications for existing WWTPs.

The next part of this thesis establishes what is meant by sustainable WWM, providing a basis for assessing the sustainability of proposed pathways. Numerous criteria and indicators for sustainability assessment have been proposed in previous research. The challenge was therefore not to identify additional criteria, but to determine which to include for an assessment to be both holistic and manageable. This thesis addressed this by proposing a definition of sustainable WWM that draws on well-established sustainable development concepts and frameworks and specifies environmental, social, and economic objectives that need to be considered simultaneously. For a solution to be sustainable in absolute terms, all sustainability objectives would need to be fulfilled. However, this is unlikely to be achievable in practice. The proposed definition instead provides a basis for assessing solutions across all relevant sustainability objectives, making it possible to identify where objectives are met, where improvements are needed, and where potential conflicts may arise. This can reduce the risk of overlooking negative consequences of improvements in one area for another. A common definition may also facilitate collaboration across groups and disciplines, allowing them to focus on different sustainability areas while remaining aware of the others. The proposed definition thus provides guidance on what to consider and strive for in sustainable WWM, as well as on selecting criteria for assessing progress towards these objectives.

Together, the future scenarios and the proposed definition of sustainable WWM provide a foundation for identifying and assessing possible pathways, with the scenarios offering opportunities to develop more robust solutions under uncertain future conditions. In this thesis, a

sustainability assessment method was used to evaluate alternative solutions to future challenges. The assessment allowed these alternatives to be systematically compared and future uncertainties considered, while identifying differences in performance and important trade-offs between sustainability objectives. These insights can support an iterative process, in which identified weaknesses inform the development and reassessment of improved solutions. Sustainability assessment can thereby support the development of more sustainable pathways, while helping to identify pathways that should be avoided.

The MCA-based sustainability assessment developed in this thesis introduced WSRs and monetary-based weighting, both based on the fact that including cost as a criterion in an MCA means that the assigned weights imply monetary values for other criteria, such as €/tCO_2e. WSRs and monetary-based weighting make these valuations visible, thereby improving transparency and providing a clearer basis for understanding weighting decisions. Linking the valuations to external reference values, including those compiled in this thesis, can further strengthen the assessment and help evaluate whether the weights are reasonable. Although subjectivity remains in the choices made, the underlying assumptions and the consequences of trade-offs become more visible and open to discussion. Sensitivity analyses can also be used to test different value preferences and changing conditions, such as different future scenarios, showing whether the preference for an alternative is robust or sensitive to particular assumptions. Stakeholders may legitimately prioritise sustainability objectives differently depending on local conditions, regulations, and organisational or political priorities. The aim is therefore not to prescribe a single correct weighting or solution, but to provide a practical and transparent procedure to support well-informed decisions. Stakeholder involvement is therefore important throughout the process. In this thesis, stakeholders were involved in both the development of future scenarios and the sustainability assessments conducted in the case studies. Such involvement can strengthen the credibility of assessments and acceptance of their results. Further refinement of the proposed definition of sustainable WWM could also benefit from the involvement of stakeholders and experts.

One limitation of this thesis is the system boundary, which focuses on WWM and mainly WWTPs. While this scope was necessary to keep the research manageable, a broader perspective covering the entire urban water system would better account for interactions between water abstraction, drinking water production, wastewater treatment, water reuse, and the receiving environment. For example, increased water reuse may reduce drinking water demand, meaning that the most sustainable pathways may require coordination between wastewater management and drinking water supply. Future research could extend the proposed framework to the entire urban water cycle to better account for system-wide interactions and sustainability implications and benefit from cross-sector collaboration.

A further consideration is that future scenarios will gradually become outdated as new information becomes available and conditions change. However, the main contribution is not the specific scenarios, but the structured approach used to develop them, which can be reapplied when needed. In many practical situations, some future conditions are already known, such as an increase in the population connected to a WWTP or stricter regulatory requirements, while others remain uncertain. In such cases, the approach can combine known conditions with plausible assumptions about uncertain developments, helping to identify solutions that are both robust and adaptable.

A challenge with monetary-based weighting is the availability of suitable external reference values. In this thesis, reference values were compiled for a subset of the criteria, but their

availability varied considerably. For climate impact, many reference values were available, whereas for BOD the basis was very limited. Establishing reference values is even more challenging for qualitative criteria, although a monetary value can still be assigned to a change of one score step. Further work is therefore needed to expand the reference-value basis and explore suitable approaches for criteria that are difficult to quantify.

Overall, this thesis demonstrates that pathways towards future sustainable WWM can be explored systematically by integrating future conditions, a holistic definition of sustainable WWM, and an assessment method within a common framework. Although the framework can be further refined, it can be applied in practice and lead to valuable insights. By making uncertainties, priorities, and trade-offs more visible, it can contribute to more informed and robust long-term decisions in WWM.

7 Conclusions and future work

This chapter summarises the main conclusions of this thesis and outlines possible directions for future work.

7.1 Conclusions

The following main contributions address the overall aim and research objectives of this thesis:

A stepwise approach for developing future scenarios for WWM, involving stakeholders and experts, was developed, progressing from identifying a broad range of future conditions to their specification and combination into quantified scenarios. The approach also identified and ranked sustainability priorities for future WWM. Expert refinement and validation increase scenario plausibility, while stakeholder involvement promotes understanding of future challenges. Using a range of future scenarios supports the development of flexible and adaptable wastewater systems. Combining quantified scenarios with process modelling enables assessment of future WWTP performance and upgrade needs.

A definition of sustainable WWM was proposed, comprising eleven sustainability areas: sanitation and hygiene, ecosystems of the recipients, recovery of nutrients, protection of biodiversity and ecosystems, substances of emerging concern, climate change, energy and resources, water reuse, cooperation at different levels, fairness and other social aspects, and economy. Each area is specified through objectives and aligned with literature-based criteria, providing a comprehensive foundation for applying the definition in practice through sustainability assessments.

A consistent, practical MCA-based sustainability assessment method was developed, which can be used together with the proposed definition of sustainable WWM and adapted to different contexts. The method introduces WSRs, which make the relative valuation of criteria explicit and the weighting process more transparent. Monetary-based weighting further allows weights to be derived from monetary valuations (e.g. €/tCO_2e or €/kWh) that can be linked to reference values. The method provides both a sustainability ranking and insights into the alternative strengths, weaknesses, and trade-offs between criteria, while sensitivity analyses help assess result robustness and account for uncertainties.

Together, these contributions provide complementary tools for supporting the selection and development of more sustainable WWM solutions.

7.2 Future work

Further development of the framework proposed in this thesis could enhance its robustness and practical applicability. Several directions for future work have been identified:

- **Further develop and validate the definition of sustainable WWM.** Application in different contexts and broader stakeholder involvement could help assess its completeness and usefulness and identify areas, objectives, or criteria requiring refinement.
- **Expand the system boundaries.** Extending the framework to a larger part of the urban water cycle could capture interactions and trade-offs between water supply, wastewater, and stormwater, although this would increase complexity and data requirements.

- **Explore non-linear scoring.** Non-linear scoring functions could better represent criteria for which equal changes in performance do not result in equal impacts. Further work would be needed on how such functions can be combined with monetary-based weighting.
- **Improve weighting of non-quantifiable criteria.** Weighting of such criteria remains challenging, and the proposed value per weighted score step was not always intuitive. Alternative approaches may be explored.
- **Compile reference values for additional criteria.** Compiling monetary reference values for additional criteria would provide a broader basis for transparent and consistent weighting.
- **Integrate complementary methods into the sustainability assessment.** Process modelling, LCA, and economic models could provide detailed quantitative input to the assessment method on environmental impacts, future costs, and the performance of alternative process configurations.

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