Wastewater treatment solutions for climate mitigation: a practical guide

Monitoring, managing and reducing greenhouse gas emissions from wastewater treatment plants

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UN Environment Programme-International Ecosystem Management Partnership (UNEP-IEMP)
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Executive Summary: Can Wastewater Treatment lead the way towards Carbon Neutrality?

The wastewater treatment sector is a substantial, yet often overlooked, contributor to the global CO_2 footprint. With an estimated contribution of 0.5–1% to global emissions, wastewater treatment is comparable to other well-known sectors, e.g., chemical manufacturing, cement production, aviation and shipping. Furthermore, it is a substantial contributor to both nitrous oxide (3–7%) and methane (7–10%) emissions globally.

Wastewater treatment has historically played a fundamental service to societies through public health and sanitation. During the past decade, the wastewater treatment paradigm has changed radically, as a result of sustainable development goals (SDGs) and climate change being prioritized in the global agenda. With these new priorities, the operational landscape for wastewater practitioners (utility managers, operators and designers) is expected to soon become increasingly complex, requiring the combination of sanitation goals with sustainability drivers – including, but not limited to, the reduction of CO_2 footprint.

Wastewater treatment plants (WWTPs) contribute to greenhouse gas (GHG) emissions directly and indirectly through:

- Fugitive emissions of nitrous oxide (N_2O) and methane (CH_4), both greenhouse gases exhibiting considerably higher global warming potential than CO_2
- Use of energy (power and heat) to support wastewater treatment operations
- Use of chemicals and materials as part of treatment processes
- Discharge or disposal of end products (treated effluent, biosolids) to the environment or other destinations
- Use of materials for construction and maintenance of WWTP infrastructure

The present technical guide, while providing an overview of the state-of-the-art knowledge on GHG emissions and contributors from a global perspective, **provides wastewater practitioners, policymakers and academics with actionable information on how to**:

- **Measure, monitor and quantify GHG emissions through existing methodologies and technology**
- **Mitigate GHG emissions through**
    - **Long-term holistic planning, and short-term optimization measures, both of which have already been shown to be effective**
    - **Policy making, regulatory and financial instruments to support and facilitate investments in mitigation actions**

Initial estimates have shown that a net zero wastewater treatment sector is a concrete possibility by combining short- and long-term actions [1]. Exploiting this potential proves even more urgent when considering that, as part of Sustainable Development Goal 6.3, a substantial increase in the WWTPs entering operation by 2030 is expected.

As a result, this technical guide highlights several recommendations for leading the transition to sustainable wastewater management and actively moving towards a net zero wastewater sector. While more information is available for centralized treatment systems, there are multiple approaches applicable for decentralized systems as well. Seven **calls to action** are listed below **for wastewater practitioners and policymakers jointly, defining a tiered approach to advance the knowledge and create the necessary conditions to implement GHG mitigation actions**:

1. **Understand:** Use existing knowledge to gather realistic estimates of GHG emissions from various WWTPs, including both centralized and decentralized systems. While the Intergovernmental Panel on Climate Change (IPCC) provides a solid methodology, experimental evidence gathered globally from WWTP-specific and nationwide campaigns should be used to inform understanding in a local context.
2. **Monitor**: If uncertainties in the GHG emission estimates persist, experimental monitoring at full-scale wastewater treatment installations should be conducted. Methodologies and instrumentation to measure and calculate direct and indirect GHG emissions are well-established, and representative facilities of a region or country's wastewater management practices should be identified.
3. **Prioritize**: Use quantitative information to identify the main GHG emissions contributors. While N_2O, CH_4 and energy-related emissions are likely to be the main contributors, their relative importance depends on many factors, e.g., geographical location and treatment process type.
4. **Aim high**: Set ambitious, yet realistic targets for GHG reductions. Use existing data from other countries or regions to define benchmarks while aligning with national CO_2 reduction goals.
5. **Facilitate:** Use existing regulatory instruments and investments, or create dedicated ones, to facilitate the fulfillment of the defined mitigation goals.
6. **Mitigate by**
    a. **Optimizing** existing treatment processes, infrastructure and equipment with relatively small and inexpensive interventions – there is a largely underexplored GHG reduction potential to be exploited.
    b. **Adopting system-oriented approaches** for planning new treatment facilities and retrofitting / upgrading existing ones, exploiting sector coupling (water-energy) while considering local conditions for implementation.

In a rapidly evolving regulatory and technological landscape, considerable experience has already been gathered on the best practices and approaches for climate change mitigation in WWTPs. In this context, high- and middle-income countries have the key role of sharing experiences with low-income countries, supporting the fulfilment of national, regional and international CO_2 reduction targets and moving towards net zero wastewater treatment objectives.

Acknowledgements

The UNEP-DHI Centre gratefully acknowledges the numerous comments, suggestions and support received from stakeholders during the production of this technical guide. In particular, we would like to thank the Global Wastewater Initiative (GWWI) and its members Mark Christensen, Kassie Morton, Daniel Ddiba, Juliet Willetts and Manzoor Qadir, as well as Andrew Crmiston, Chao Fu, Dhammika Leshan Wannigama, Erol Cavus, Ludwig Hermann, Mona Abdulghani and Pablo César Morales Rico for the input provided. We hope the strong collaboration that shaped this guide will continue in the implementation of sustainable wastewater management and systems globally.

List of acronyms and abbreviations

A/O	Anoxic/Oxic
A^{2}O	Anaerobic/Anoxic/Oxic
AGS	Aerobic Granular Sludge
AOA	Ammonium Oxidizing Archaea
AOB	Ammonium Oxidizing Bacteria
BNR	Biological Nutrient Removal
CapEx	Capital Expenditure
C/N	Carbon-to-Nitrogen Ratio
CAS	Conventional Activated Sludge
CCAC	Climate and Clean Air Coalition
CCM	Crab Carapace Micropowder
CDM	Clean Development Mechanism
CHP	Combined Heat-Power
CNG	Compressed Natural Gas
COD	Chemical Oxygen Demand
COD/N	COD-to-Nitrogen Ratio
CW	Constructed Wetlands
DO	Dissolved Oxygen
EF	Emission Factor
EPA	Environmental Protection Agency
ESR	Effort Sharing Regulation
EWS	eThekwini Water and Sanitation
FA	Free Ammonia
FNA	Free Nitrous Acid
GC	Gas chromatography
GHG	Greenhouse Gas
GHGRP	Greenhouse Gas Reporting Program
GLAM	Guidance for LCA indicators and Methods
GMP	Global Methane Pledge
GMI	Global Methane Initiative
GWOPA	Global Water Operator Partnership Alliance
GWP	Global Warming Potential
IPCC	Intergovernmental Panel on Climate Change
IR	Infrared
ISO	International Standards Organization
LCA	Life Cycle Assessment
LNG	Liquified Natural Gas

MLE	Modified Ludzack-Ettinger
MLSS	Mixed Liquor Suspended Solids
MTDM	Mobile Tracer Gas Dispersion Method
NbS	Nature-Based Solution
ND	Nitrification-Denitrification
NDC	National Determined Contribution
NGER	National Greenhouse and Energy Reporting
NiR	Nitrite Reductase
NOB	Nitrite Oxidizing Bacteria
NOR	Nitric Oxide Reductase
N_2OR	Nitrous Oxide Reductase
OD	Oxidation Ditch
OpEx	Operational Expenditure
PACM	Paris Agreement Crediting Mechanism
PE	Population Equivalent
PHA	Polyhydroxyalkanoate
PNA	Partial Nitritation Anammox
PV	Photovoltaic
RNG	Renewable Natural Gas
SBR	Sequencing Batch Reactor
SCADA	Supervisory Control and Data Acquisition
SDG	Sustainable Development Goals
SISs	Soil Infiltration Systems
MBR	Membrane BioReactor
SDG	Sustainable Development Goal
THP	Thermal Hydrolysis Process
TN	Total Nitrogen
UD	Urine-diverting
VFD	Variable Frequency Drive
WRRF	Water Resource Recovery Facility
WSP	Waste Stabilization Ponds
WWTP	Wastewater Treatment Plant

List of case studies

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1. Introduction

Greenhouse gas (GHG) emissions are a global concern due to their significant impact on the Earth's atmosphere, resulting in global warming and climate change. Climate change affects all facets of life and causes extreme weather events, droughts, water quality issues, and flooding. An international, cross-sector response is required to reduce global GHG emissions and abate further adverse impacts. The wastewater sector – the third-largest source of nitrous oxide (N_2O) and the fifth-largest source of methane (CH_4) emissions globally – is no exception [2].

It has been estimated that the wastewater management sector is responsible for 3-4% of the energy use and 1-3% of GHG emissions globally [5, 6], with a considerable contribution from wastewater treatment (0.5-1%). While emissions from this sector have been small relative to those from other sectors, recent trends point to increases in relative contributions globally, resulting from both the wastewater treatment plants (WWTPs) already in operation and particularly the construction of new facilities in non-sanitated areas [3]. The choice of low-carbon technologies is therefore crucial for a sustainable implementation of sanitation goals at a global level.

Wastewater treatment contributes to GHG emissions directly and indirectly, in three main "scopes" defined by the international GHG Protocol:

> **Scope 1** involves direct emissions from wastewater facilities and utility operations, such as CH_4 and N_2O emissions from treatment processes, fossil-based carbon dioxide (CO_2) emissions from vehicles and diesel generators, CH_4 emissions from onsite sludge storage and disposal, and CH_4 emitted during biogas or natural gas combustion.
>
> **Scope 2** consists of indirect emissions from purchased fossil-based energy sources used for on-site processes or operations (e.g., electricity, heat, or steam).
>
> **Scope 3** covers indirect emissions from supply chains, including construction materials, equipment and chemicals used for treatment processes, as well as end products, such as treated wastewater discharges, untreated wastewater overflows, and offsite sludge disposal.

While the Intergovernmental Panel on Climate Change (IPCC) has developed detailed guidelines for the determination of GHG emissions from wastewater, it should be noted that mitigation activities are not commonly reported under National Determined Contributions (NDCs) [4].

Traditionally, WWTPs have focused on complying with increasingly stringent effluent discharge regulations and implementing appropriate resource recovery strategies. In recent decades, wastewater utilities have used process intensification, expansion, new technologies, and optimization methods to meet their goals. Now, they are also required to reduce their GHG footprint and operate at net zero emissions. These competing priorities expose WWTPs to a unique set of challenges that demand holistic decision-making during both design and operations. Yet, connecting climate actions with sustainable development goals (SDGs) has the potential to unlock additional sources of financing to support wastewater infrastructure and development projects. Thus, the involvement of utility managers, process engineers and operators is necessary to guide wastewater-sector NDCs and achieve decarbonization goals.

This technical guide aims to provide wastewater professionals and decision-makers with the knowledge and tools to be able to identify their direct and indirect GHG emissions, select and implement comprehensive mitigation actions, and adopt appropriate design aspects and criteria and operational procedures and guidelines. The guide is also expected to provide policymakers with the relevant background information required to define local and regional policy-making actions to support GHG mitigation and sustainable implementation plans for new WWTPs. Although not specifically addressed to academics, this guide also provides an overview of the state-of-the-art knowledge on GHG emissions in WWTPs, thereby guiding future research.

Through a systematic review of scientific articles, databases, published reports and case studies, the guide aims to:

- provide a comprehensive evaluation of GHG emissions from wastewater treatment practices on a global scale,
- identify, through use cases and technology assessments, key GHG reduction actions for relevant stakeholders in both planning and optimization,
- advocate for prioritizing GHG mitigation in the wastewater sector to address climate change.

Within the wastewater management cycle, the technical guide will specifically focus on **wastewater treatment and resource recovery**, including centralized and decentralized wastewater treatment and management of sludge.

2. GHG emissions in the wastewater management cycle: An overview

Greenhouse gas (GHG) emissions from wastewater management systems are increasingly recognized as a significant contributor to the overall carbon footprint of urban infrastructure. A comprehensive understanding of these emissions requires not a focus on direct emissions from treatment processes, but also on the energy consumed, chemicals used, and the broader supply chain impacts.

This section provides an overview of how emissions are classified and quantified in line with IPCC Guidelines for National Greenhous Gas Inventories and the GHG Protocol's Scope 1, 2, and 3 frameworks. Together, these approaches allow both utilities and national authorities to account for emissions consistently while supporting targeted mitigation planning.

2.1. Classification of GHG emissions

Within the IPCC Guidelines, emissions from wastewater are categorized under the Waste sector. Primary direct emissions include methane (CH_4) and nitrous oxide (N_2O), generated through biological transformation processes and the decomposition of organic matter. By contrast, direct carbon dioxide (CO_2) emissions from wastewater are not included in national GHG totals, as these are generally derived from natural (biogenic) organic matter in human excreta or food waste.

The IPCC defines three methodological tiers for estimating wastewater emissions:

- **Tier 1**: Applies global default emission factors (EFs) and basic activity data. Suitable for high-level national reporting, it lacks detail for site-specific mitigation planning.
- **Tier 2**: Uses country- or system-specific factors, offering improved accuracy for regions with detailed datasets or studies.
- **Tier 3**: Relies on direct measurements or advanced modeling, enabling utilities to capture real-time dynamics and operational variability for targeted mitigation.

These tiers guide the choice of monitoring strategy based on the availability of data, resource capacity, and the goals of the emissions inventory. Tier 1 provides a valuable starting point using default EFs based on influent characteristics and treatment type. As data collection capabilities improve, progression to Tier 2 and 3 is encouraged to support more precise emissions management.

As an example, for N_2O the 2019 refinement of the IPCC Guidelines introduced an updated Tier 1 default EF of 1.6% of influent total nitrogen load emitted as N_2O-N (0.016 kg N_2ON per kg TN) for conventional WWTP. This revision reflects improved understanding of biological nitrogen removal processes and recognizes that earlier estimates (0.05% in the 2006 Guidelines) substantially underestimated emissions. While useful for countries with limited data, this default value still represents a generalized approximation and does not account for the wide variability in treatment technologies and operational practices.

While the IPCC tiers describe how emissions are estimated, the GHG Protocol's Scope 1, 2 and 3 framework describes where emissions occur within the utility's sphere of influence. Scope 1 covers direct process and energy emissions from treatment units and on-site systems, Scope 2 covers indirect emissions from purchased electricity, and Scope 3 covers upstream and downstream emissions across the value chain. An overview of Scope 1, 2 and 3 emissions in a conventional centralized WWTP is provided in Figure 2-1, highlighting the most prominent sources of emissions. The following subsections detail these categories in the context of wastewater management, providing the foundation for understanding both the cause of emissions and potential pathways for mitigation.

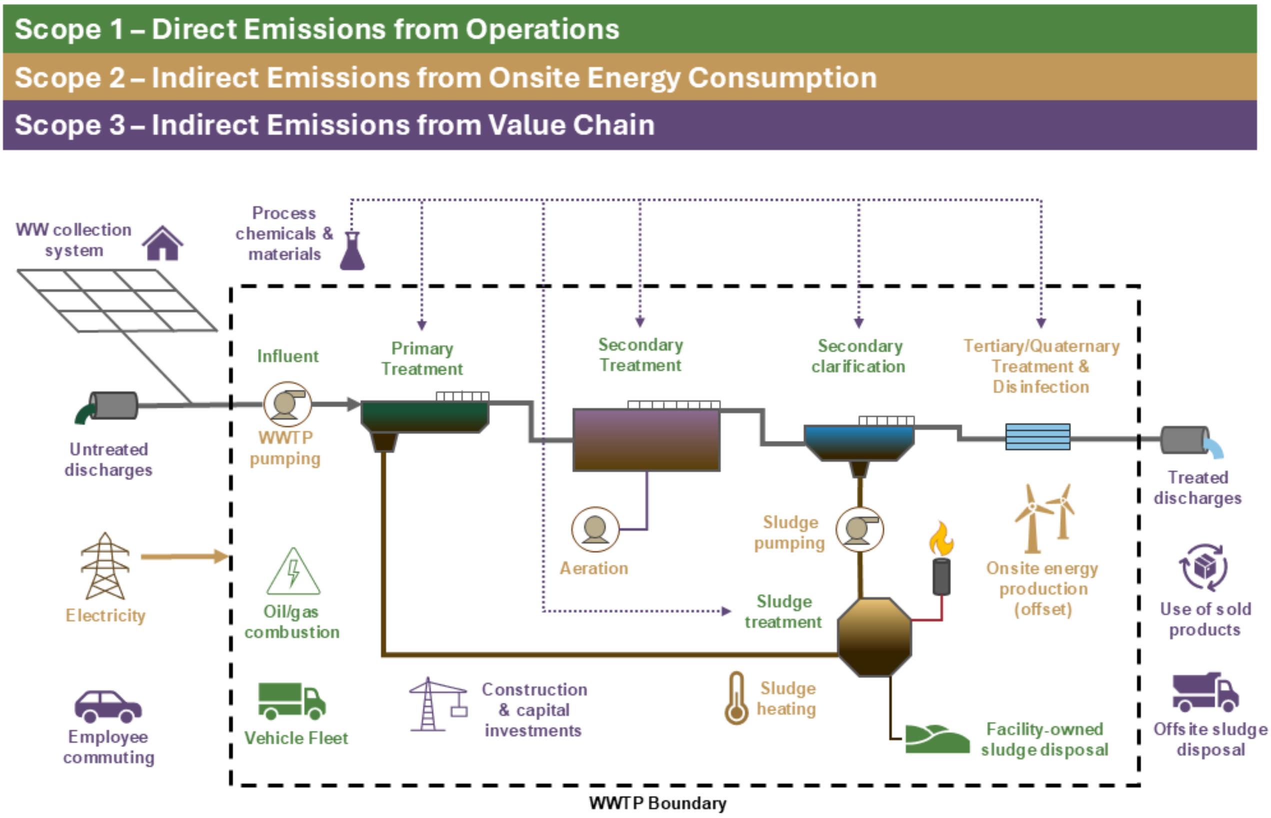

Figure 2-1. Overview of common GHG emissions from wastewater treatment plants (WWTPs).

2.1.1. Scope 1 emissions

Scope 1 emissions include all direct greenhouse gas releases from treatment processes. In WWTPs, these are primarily methane (CH_4) and nitrous oxide (N_2O) from treatment units and on-site energy systems and can generally be controlled by wastewater operators through process selection and optimization. While carbon dioxide (CO_2) is also produced during biological treatment through natural respiration, its contribution is considered predominantly biogenic and has an insignificant impact compared with CH_4 and N_2O.

Nitrous oxide (N_2O)

N_2O is a highly potent greenhouse gas with a global warming potential (GWP) over a 100-year period 273 times greater than carbon dioxide (CO_2) [5]. It persists in the atmosphere for 109 years and is the most important factor in stratospheric ozone depletion, while also indirectly affecting human health (e.g., by increasing the occurrence of skin cancer) [5, 6].

N_2O is released from multiple anthropogenic sources, including soil management, fuel combustion and industrial processes. Within this spectrum, wastewater treatment is recognized as the most significant non-agricultural source accounting for an estimated 3-7% of global anthropogenic N_2O emissions [7]. From 1980 to 2015, emissions from the wastewater sector have been estimated to have increased steadily at a rate of approximately 0.04 Tg N_2O-N per year [8]. This trend is projected to continue, driven by the growing implementation of biological nitrogen removal processes to comply with increasingly stringent effluent standards, and the expansion of wastewater infrastructure to serve growing populations and accelerating urbanization.

Although the microbial pathways responsible for N_2O formation are relatively well-documented, the emissions themselves depend heavily on operational conditions, wastewater characteristics, and environmental factors. N_2O is first formed in the liquid phase and then transferred into the atmosphere through mechanisms such as stripping in aerated zones or natural diffusion in non-aerated systems [9]. Emissions vary significantly based on operational parameters such as dissolved oxygen levels, carbon-to-nitrogen ratios, and transient loadings. Advanced Tier 3 monitoring approaches, e.g. online monitoring of liquid-phase or gas-phase N_2O, help identify high-emission phases and identify mitigation strategies.

The 2019 Refinement for the 2006 IPCC Guidelines offers a framework for estimating N_2O emissions from domestic wastewater. For countries with limited data, a Tier 1 approach using default EFs (based on per capita nitrogen load and treatment system types) can provide a baseline estimate. This method assumes typical operating conditions and does not require site-specific data. Where more data are available, Tier 2 and 3 approaches allow for inclusion of country-specific nitrogen content in wastewater, treatment technology distribution, and measurement-based emissions factors.

Methane (CH_4)

CH_4 is a potent GHG with a GWP 28 times greater than that of CO_2 over a 100-year period [5]. The wastewater sector contributes approximately 7-10% of global anthropogenic methane emissions [10]. Methane is primarily produced in anaerobic processes, including sludge digestion for biogas production and onsite fecal sludge management. These processes involve sequential microbial steps under oxygen-free conditions, converting biodegradable organic carbon into biogas – a mixture of CH_4 (60–70%), CO_2 and traces of H_2S.

Due to the high organic content of wastewater, unintentional methane production also occurs in sewer networks, especially in anaerobic zones within pipes carrying raw sewage. The CH_4 generated in these zones is often emitted into the atmosphere through turbulence-induced stripping in the pipelines, leading to direct and diffuse emissions, and during preliminary treatment steps in WWTPs (headworks, screening and grit removal).

Within WWTPs, the predominant methane source is anaerobic sludge management. Biogas produced during digestion is typically used for heat and power generation via gas engines. However, notable emissions can still occur as a result of:

- Fugitive emissions from digesters, pressure relief valves, and poorly sealed tanks,
- Gas leaks in piping or joints,
- Open-air storage of (un)digested sludge, where residual organic matter continues to produce methane pre- or post-digestion.

For countries with limited resources or baseline data, the IPCC Tier 1 methodology can be applied to estimate methane emissions. This involves using default EFs based on treatment system type (e.g., anaerobic lagoons, digesters) and wastewater characteristics. IPCC defaults are also available for onsite treatment systems (e.g. septic tanks, latrines) yet based on a small number of installations. While less accurate than direct measurement approaches, the Tier 1 method supports initial national-level reporting and identification of mitigation opportunities.

2.1.2. Scope 2 emissions

Scope 2 covers indirect emissions from purchased electricity and heat used onsite. These emissions depend heavily on treatment technology, process intensity, and the carbon intensity of the electricity grid. Hence, Scope 2 emissions are partially in control of wastewater operators but are also driven by local or regional energy market and policies.

Energy use in WWTPs is dominated by two key systems:

- Aeration systems: Account for 40-60% of energy use. Upgrades such as fine-bubble diffusers, real-time oxygen control, and high-efficiency blowers can reduce energy demand significantly.
- Pumping systems: Optimization of pump sizing, installation of variable frequency drives (VFDs), and maintenance scheduling contribute to energy savings.

A key influential factor in Scope 2 emission is the regional variability of the CO_2 intensity of energy used. For instance, in Chicago natural-gas is the primary energy source (269 kg CO_2 equivalent per MWh) for municipal wastewater treatment, resulting in Scope 2 emissions that are 12 times higher per volume treated than in Toronto, which relies on a low-carbon energy portfolio (22 kg CO_2 equivalent per MWh) [2]. Differences can be even greater, such as in Europe, where CO_2 intensity ranges from 8 (Sweden) to 594 (Poland) g CO_2eq/kWh [11]. Although WWTPs cannot always choose their primary energy source, they can still influence their Scope 2 emissions by reducing energy consumption, improving energy recovery, and incorporating renewable energy sources. Specific measures include:

- Heat recovery from digestate or process water (e.g., via heat pumps or exchangers), supporting plant heating needs or being exported to the district heating network.
- Electricity and heat generation through combined heat and power (CHP) systems fueled by biogas, providing a renewable energy source that can be used onsite or exported to the grid.
- Integration of renewable sources, such as on-site solar photovoltaic systems, wind energy systems, and procurement of green electricity.

Overall, decarbonization of the energy sector has the potential to reduce Scope 2 emissions from the wastewater sector by up to 59% globally over the next 10 years [2].

2.1.3. Scope 3 emissions

Scope 3 emissions include all other indirect emissions that occur in the value chain but are outside the direct control of the utility or owner of the treatment facility. Generally, Scope 3 categories include, e.g. purchased and capital goods, fuel- and energy-related activities, upstream transport and distribution, waste generated in operations, downstream transportation and distribution, end-of-life treatment of sold products. In WWTPs, these often make up large portions of a utility's total emissions, whereas the relative contribution may be reduced in decentralized or onsite treatment systems, depending on the type of installation. The most relevant Scope 3 emissions include:

- process chemicals and materials
- off-site emissions (upstream and downstream)
- construction-derived emissions.

Process chemicals inputs such as methanol for external carbon dosing, phosphorus precipitants (e.g., ferric chloride), and polymers have embedded emissions associated with their manufacture and transport. Activated carbon used in advanced treatment also has a considerable carbon footprint due to its energy-intensive production and regeneration process.

Off-site emissions originate from:

- Sludge disposal, e.g.,
    - Land application can result in N_2O emissions depending on nitrogen content and soil conditions
    - Landfilling releases methane unless gas capture is implemented
    - Incineration produces CO_2 and potentially N_2O, especially from biosolids with high nitrogen content.
- Treated and untreated discharges, including overflows or bypasses: Nitrogen discharged in effluent can convert to N_2O in receiving waters, while remaining COD can result in CH_4.

Emissions associated with construction and infrastructure upgrades include material extraction, manufacturing, and transport. For example, concrete and steel have different embodied emissions, e.g., steel often has higher emissions per ton, but structural performance and service life must also be considered. A recent example from Ejby Mølle WWTP (Denmark) showed that construction choices (e.g., use of steel rather than concrete for a sludge storage tank) can reduce CO_2 emissions (in this case, 7 kg CO_2-eq savings per m^3 of stored sludge) [12].

Key take-aways:

- GHG emissions are categorized as Scope 1 (direct), 2 (energy-related), and 3 (indirect upstream/downstream emissions)
- Nitrous oxide (N_2O) and methane (CH_4) are the dominant direct GHG emissions from wastewater processes
- IPCC’s Tier 1–3 methods differ by data needs and accuracy
- Updated emission factors show both N_2O and CH_4 emissions are more significant than previously estimated

2.2. Causes and mechanisms for Scope 1 emissions

Scope 1 emissions are direct GHG emissions that occur during biological treatment processes. The primary GHGs are methane (CH_4) and nitrous oxide (N_2O), with carbon dioxide (CO_2) also emitted through microbial respiration. While CO_2 is naturally produced as organic carbon is oxidized during wastewater treatment, it is generally considered biogenic in origin. However, the 2019 IPCC Refinement suggests quantifying CO_2 emissions when there is a direct way to measure the contributions of non-biogenic, fossil fuel-derived organic carbon or a suspicion of its presence in wastewater [13]. This approach helps avoid potential underestimation in sanitation systems, where differentiated infrastructure for industrial and municipal wastewaters is often lacking [14]. In the municipal context, direct CO_2 emissions contribution to the net carbon footprint of wastewater treatment is insignificant compared to CH_4 and N_2O. Thus, CH_4 and N_2O are the dominant gases of concern due to their high global warming potentials (GWP) of 27–30 and 273 times that of CO_2, respectively [5].

2.2.1. Nitrous oxide (N_2O)

N_2O emissions arise mainly in biological reactors from the microbial transformation of nitrogen during wastewater treatment, specifically nitrification (aerobic oxidation of ammonium to nitrite and nitrate) and denitrification (anoxic reduction of nitrate/nitrite to nitrogen gas) processes. N_2O is produced as a byproduct or intermediate when these pathways are incomplete or imbalanced. After formation in the liquid phase, N_2O is transferred to the atmosphere either through air stripping in aerated zones or diffusion in non-aerated systems.

Several interrelated factors influence the magnitude and viability of N_2O emissions:

- ***Aeration regimes and dissolved oxygen (DO) concentrations:*** Uneven or intermittent aeration can cause local DO fluctuations, creating transient conditions favourable for N_2O release. Low DO concentrations favour incomplete nitrification and nitrite accumulation, while excessive DO concentrations suppress denitrification, both of which enhance N_2O production. In naturally aerated systems, where DO concentration is not regulated, an increase in carbon (i.e. chemical oxygen demand, COD) or NH_4 loading leads to an increase in oxygen consumption and unstable nitrification, resulting in increased N_2O production.
- ***Nitrite accumulation:*** N_2O production has been hypothesized to be strongly linked to nitrite (NO_2^-) build-up, which occurs when nitrification is interrupted (e.g., low dissolved oxygen, toxic shocks) or when denitrification lacks sufficient carbon to proceed to completion. Elevated NO_2^- levels provide a substrate for both nitrifier-denitrification and incomplete heterotrophic denitrification, enhancing N_2O emissions.
- ***Sludge age:*** In activated sludge systems, sludge age governs microbial community balance. Low sludge age favours faster-growing ammonium oxidizers but can wash out nitrite oxidizers, causing NO_2^- accumulation and N_2O peaks. In ponds, lagoons, and septic tanks, sludge depth and accumulation create stratified zones where incomplete nitrification-dentification cycles promote localized N_2O production.
- ***pH:*** pH influences both nitrification and denitrification. Nitrifiers operate optimally around 7.5–8.0, whereas denitrifiers prefer near-neutral conditions (6.5–7.5). Acidic conditions (<6.5) inhibit denitrification, often leading to N_2O accumulation, while alkaline conditions shift the equilibrium of dissolved nitrogen species.
- ***Temperature:*** Higher temperatures increase microbial metabolism and the rate of ammonia oxidation. This accelerates nitrification and the potential for nitrite

accumulation if not matched with adequate denitrification capacity. Consequently, N_2O emissions may rise in warm climates or during summer operation, unless counterbalanced by sufficient organic carbon and DO control.

- ***C/N ratio:*** C/N ratio is critical for complete denitrification. When organic carbon is limited, denitrification stalls at nitrite or N_2O instead of proceeding to N_2. Conversely, higher C/N ratios support full denitrification but may increase CH_4 and CO_2 production in anaerobic zones. Balancing the influent C/N ratio (often through supplemental carbon dosing) is therefore a central mitigation strategy.
- ***Ammonium oxidation rate:*** High ammonium oxidation rates, particularly under transient DO conditions, are associated with elevated N_2O production by autotrophic nitrifiers. This link underscores the importance of matching aeration and nitrification capacity to influent ammonia loads

These factors are further elaborated upon in the following sections.

2.2.2. Methane (CH_4)

CH_4 emissions in wastewater treatment arise primarily from anaerobic biological processes and fugitive losses during sludge storage, digestion, and handling. Methane is produced by methanogenic archaea under anaerobic conditions through the degradation of organic substrates, typically acetate, hydrogen, and carbon dioxide. In the WTTP, methane emissions can also occur at inlet or in initial treatment steps (headworks, screening, grit removal, primary clarifiers), following methane generation in upstream sewer networks, depending on operational conditions and system design.

Key mechanisms and influencing factors affecting methane production include:

- ***Redox conditions:*** Methanogenesis is strictly anaerobic. Oxygen exposure inhibits methanogens, inhibiting CH_4 production or oxidizing CH_4 to CO_2. Anaerobic zones exist naturally in septic tanks, Imhoff tanks, sludge digesters, waste stabilization ponds, and even in anoxic niches within activated sludge flocs. CH_4 formed in these zones may dissolve in the liquid phase and be stripped during aeration or released directly to the atmosphere.

- ***pH:*** Methanogens are highly sensitive to pH, with an optimal range around 6.8–7.4. Acidic conditions (<6.5) inhibit CH_4 production and favour volatile fatty acid accumulation, while alkaline conditions (>8.2) reduce methanogen viability and shift biogas composition towards CO_2.

- ***Retention time:*** Sufficient hydraulic and solids retention time (HRT and SRT) are critical for methanogenesis. Methanogens have slow growth rates, and retention times of 10–

30 days are typically required for stable CH_4 production in anaerobic digesters. Short-circuiting in septic systems or excessive sludge withdrawal can reduce conversion efficiency and increase CH_4 dissolved in effluent.

- ***Temperature***: Temperature strongly influences methanogenic activity. Optimal methanogenesis occurs in the mesophilic range (30–38 °C) and is further accelerated under thermophilic conditions (50–55 °C). In anaerobic digesters, the CH_4:CO_2 ratio in biogas typically ranges from 60:40 to 70:30 at optimal temperatures. At lower temperatures (<20 °C), methanogenesis slows substantially, leading to incomplete COD conversion and reduced CH_4 yields. Seasonal fluctuations and the amount of infiltration and inflow to the sewer system affect wastewater temperatures, so the ability to control temperature prior to anaerobic digestion process is a crucial at conventional WWTPs. However, low-technology or nature-based wastewater treatment installations are fully reliant on incoming sewage temperature and lack external heating sources. In cold climates, on-site systems, such as septic tanks and latrines, may experience periods of very low temperatures or even freezing, during which biological activity ceases and emissions are negligible. Meanwhile, in tropical and subtropical regions, particularly in open systems such as stabilization ponds, temperature variations play a critical role in methane dynamics. Evidence suggests that emissions in these climates are strongly modulated by seasonal temperature changes and photoperiods [15-18].

Collectively, these factors determine how much of the influent organic load is converted into CH_4. While engineered anaerobic digesters can capture and utilize biogas as an energy source, uncontrolled anaerobic processes (e.g., septic tanks, ponds, sewer networks) often release CH_4 directly to the atmosphere, contributing significantly to Scope 1 emissions.

Key take-aways:

- N_2O emissions arise from imbalanced nitrification–denitrification, strongly influenced by DO control, nitrite build-up, sludge age, pH, temperature, and C/N ratio.
- CH_4 emissions stem from anaerobic zones in sewers, tanks, and sludge systems, shaped by redox conditions, pH, retention time, and temperature.
- Operational instability and uncontrolled anaerobic processes significantly increase Scope 1 emissions, whereas well-managed biological treatment and digesters can reduce nitrous oxide emissions and capture methane for energy use, respectively

2.3. GHG Emissions from Centralized Systems

Centralized WWTPs represent the backbone of modern urban sanitation and are designed to remove organic matter, nutrients, and pathogens in wastewater collected by urban sewer systems serving high-density communities before effluent is discharged into the receiving environment. While these systems provide substantial water quality and public health benefits, they are also an important source of GHG emissions, contributing to the overall carbon footprint of the water sector. Owing to easier accessibility for process evaluation and monitoring, GHG emissions from centralized WWTPs are relatively well understood.

Field measurements from centralized WWTPs reveal that N_2O emissions can differ considerably from the IPCC 2019 default EF of 1.6% of influent total nitrogen (TN). For example, recent studies have highlighted elevated N_2O emissions from multiple WWTPs, with some plants exceeding 8% of the influent total nitrogen load [19, 20]. Within a single plant, N_2O emissions can account for as much as 86% of total direct GHG emissions (Scope 1 emissions) based on actual monitoring data [21, 22]. These findings underscore the vulnerability of centralized systems to operational conditions that favour incomplete nitrification or denitrification, making them hotspots for N_2O generation. Plant-wide emissions studies show that while bioreactors are often the dominant source of N_2O (81-99.8%) [23, 24], other units such as primary clarifiers and sludge treatment units can also be major contributors, accounting for up to 87% of plant-wide N_2O emissions [25]. This highlights the importance of adopting a whole-plant perspective rather than focusing exclusively on biological reactors.

Methane (CH_4) is another critical component of centralized systems' GHG profile, originating mainly from anaerobic digestion of sludge and subsequent handling of digested solids. Although many WWTPs now capture and utilize biogas for combined heat and power (CHP) or upgrading to biomethane, fugitive losses remain a concern. Methane can escape through leaks in digesters, piping systems, or open storage tanks, as well as from uncontrolled anaerobic zones in sewer networks and treatment basins. The extent of these fugitive emissions varies widely, depending on infrastructure integrity, operational practices, and the degree of gas recovery and utilization.

As electricity grids continue to decarbonize and WWTPs increasingly adopt renewable energy solutions such as solar, wind and advanced energy recovery from biogas, Scope 2 emissions are projected to decline. In this context, Scope 1 process emissions (CH_4 and N_2O) are expected to dominate GHG profiles of centralized systems in the coming decades. This shift places increased urgency on developing a deeper understanding of emission pathways, identifying process conditions that drive GHG formation, and implementing operational and technological strategies for mitigation. Nevertheless, the relevance of Scope 2 and Scope 3 emissions will also be discussed in the guide.

2.3.1. Direct emissions from centralized systems

The following sections examine in more detail the role of process type and operational conditions in influencing N_2O and CH_4 emissions and the influence of operational conditions, providing a foundation for identifying targeted interventions to reduce Scope 1 emissions in centralized treatment plants.

The role of process types on nitrous oxide (N_2O) emissions

The configuration of treatment processes is one of the most important determinants of N_2O emissions from centralized wastewater treatment plants. Different biological nitrogen removal (BNR) pathways create distinct operational environments that shape the microbial transformations of nitrogen and, consequently, the likelihood of N_2O generation. While the IPCC 2019 Refinement recommends a Tier 1 default EF of 1.6% of influent TN as N_2O-N, recent large-scale assessments demonstrate that actual emissions can vary widely across process types. A comprehensive meta-analysis compiled 376 emission factor observations from more than 200 facilities worldwide, significantly expanding the empirical basis for wastewater-related N_2O [26]. This dataset includes both individual bioreactor-, side stream processes and plant wide-level measurements, and identified potential trends (Figure 2-2) while revealing high variability across biological treatment processes.

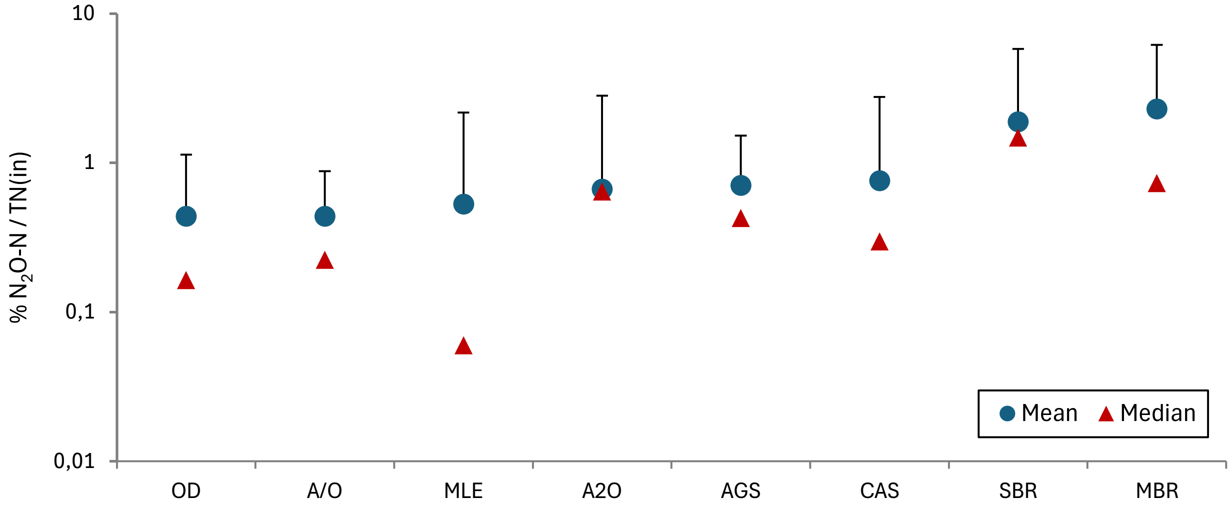

Figure 2-2. N_2O emission factors (arithmetic mean, median and standard deviation of literature studies) associated with different biological treatment processes. The presented data are adapted from [26] and include technologies, for which only more than 10 data sets were available. OD: Oxidation Ditch, A/O: Anoxic/Oxic, MLE: Modified Ludzack-Ettinger, A^2O: Anaerobic/Anoxic/Oxic, AGS: Aerobic Granular Sludge, CAS: Conventional Activated Sludge, SBR: Sequencing Batch Reactor, MBR: Membrane BioReactor.

Importantly, the spatial scale of measurement strongly influences emission estimates. Clear discrepancies were demonstrated [26] between EFs reported at the bioreactor scale, the plantwide scale, and for sidestream treatment. While bioreactor measurements show the widest variability (0.00003–20.69% kg N_2O-N kg^{-1} TN) and confirm these units as emission hotspots, plantwide campaigns indicate that other unit processes – such as primary or sludge treatment – can also dominate emissions, contributing up to 87% of total plantwide N_2O in some cases [26]. This highlights that using bioreactor-specific EFs to represent whole-plant emissions may systematically underestimate total emissions and underscores the importance of scale-differentiated data for accurate inventories.

Based on the magnitude of N_2O emissions, all treatment processes can be roughly divided into two categories, i.e. low-emission processes, with mean <1% and median <0.5% kg N_2O-N kg^{-1} TN (Figure 2-2), and high-emission processes, with mean ≥1% and median ≥0.5% kg N_2O-N kg^{-1} TN (Figure 2-2). Importantly, all the high-emission processes are designed for BNR, which is consistent with their higher emission potential. Among these, sequencing batch reactors (SBRs) and membrane bioreactors (MBRs) exhibit mean EFs that are 2–4 times higher than other BNR configurations such as anaerobic/anoxic/oxic (A^2/O) and modified Ludzack–Ettinger (MLE) processes. By contrast, systems such as A/O, MLE, and A^2/O consistently show lower emissions, often comparable to conventional activated sludge (CAS). This indicates that advanced nutrient removal does not necessarily lead to higher N_2O release, and that process design and influent characteristics play a key role in shaping emissions.

The elevated EFs observed in high-emission groups can be attributed to multiple interacting drivers, including the presence of additional anoxic and oxic stages, low influent COD/N ratios reducing denitrification capacity, elevated salinity combined with low DO concentrations, and higher nitrogen loads. These findings emphasize the need for targeted operational control and process optimization to mitigate emissions from centralized treatment.

Membrane BioReactors (MBRs) combine activated sludge with membrane filtration, enabling long sludge ages and high effluent quality, but also creating unique oxygen transfer dynamics. N_2O emissions in MBRs are influenced by a combination of operational, biological, and physical factors common to these systems:

- ***Long Sludge Retention Time (SRT) and high Mixed Liquor Suspended Solids (MLSS)*** concentration enhance microbial diversity and nitrification rates but can create localized oxygen-limited zones, where N_2O production is promoted.
- ***Membrane Fouling:*** Driven by soluble microbial products (SMP) and extracellular polymeric substances (EPS), fouling can reduce oxygen transfer efficiency and alter microbial metabolism, often leading to incomplete denitrification and increased N_2O emissions.
- ***Variable Dissolved Oxygen (DO) concentration:*** Changes in DO levels can have contradictory effects, where even higher DO concentrations might exacerbate, rather than reduce, N_2O emissions. Sudden increases in airflow can release N_2O from the liquid phase into the atmosphere.

Sequencing Batch Reactors (SBRs) operate in time-sequenced cycles rather than continuous flow, exposing microbial communities to shifting conditions that strongly influence nitrogen transformation. SBR operations consist of fill, react, settle, decant, and idle phases, each influencing carbon availability and microbial activity:

- **Low C/N Ratio** contributes to incomplete denitrification and increased N_2O production. Organic carbon may be depleted during aerobic oxidation by heterotrophs during the react phase, leaving insufficient carbon for denitrification later in the cycle.
- **Nitrite Accumulation:** Sudden changes in DO concentrations can promote nitrite build-up, a known precursor to N_2O production [27].

Hence, understanding the operational cycles and carbon availability dynamics in SBRs is critical for minimizing N_2O emissions.

The role of operational conditions on nitrous oxide (N_2O) emissions

The operation of WWTPs – and particularly their biological treatment units – plays a critical role in determining the magnitude and variability of N_2O emissions. While treatment processes set the stage for microbial transformations, localized conditions within the reactor environment often dictate whether these transformations proceed towards complete nitrogen removal or are diverted towards N_2O accumulation and release.

A key challenge is that the relationship between operational controls and N_2O emissions is not universal. Instead, it is shaped by a complex, nonlinear interplay among process parameters, microbial community structure and environmental drivers. Even within the same process configuration, emissions can vary widely depending on influent characteristics, seasonal changes and transient loadings. Despite this complexity, empirical evidence and field experience have revealed several consistent trends that help identify conditions associated with either increased or reduced N_2O formation. However, despite this complexity, certain patterns and trends have emerged from research and operational experience, revealing "low-hanging fruit" interventions that are generally effective in minimizing emissions. In this section, we will:

1. Highlight key factors influencing N_2O emissions.
2. Provide actionable insights on how engineers and operators can adjust operational parameters to create favourable local conditions for minimizing emissions.

Importantly, this section focuses on local conditions within bioreactors – such as dissolved oxygen (DO) concentration, carbon-to-nitrogen (C/N) ratio, nitrite accumulation, transient dynamics, and environmental factors like temperature and pH – that reflect the direct environment experienced by microorganisms. These differ from operational parameters (e.g., aeration rates, recirculation flows, chemical dosing), the adjustable levers used by plant operators and process engineers to influence those conditions. Understanding how local conditions shape N_2O production provides the foundation for developing effective operational strategies and targeted mitigation measures.

Dissolved oxygen (DO). Dissolved oxygen concentration is one of the most important parameters controlling N_2O emissions during both nitrification (nitratation/nitritation) and denitrification. Its effects are complex because DO influences both the pathways of N_2O production and the extent to which the gas is stripped from the liquid phase. During nitrification, low DO concentrations (<1 mg/L) are consistently associated with elevated N_2O emissions due to the activation of the nitrifier denitrification pathway. For example, Zheng et al. (1994) observed enhanced N_2O release at DO levels below 1 mg/L [28]. Li et al. (2015) reported similar observations in a pilot-scale SBR that decreasing DO from 3.0 to 0.5 mg/L led to nitrite accumulation and increased N_2O production through nitrifier denitrification [29]. A meta-analysis by Vasilaki et al. (2020) further showed a strong negative correlation (Spearman's $\rho = -0.7$) between DO concentration and dissolved N_2O emissions [30]. At moderate to high DO levels (0.2–3 mg/L), the contribution of nitrifier denitrification to N_2O decreases, while the hydroxylamine oxidation pathway becomes more dominant. Peng et al. (2014) demonstrated that when DO increases from 0 to 3 mg/L, the specific N_2O production rate rose from 0 to 1.9 mg N_2O-N h^{-1} g^{-1} VSS, with the relative contribution of nitrifier denitrification by ammonia-oxidizing bacteria (AOB) dropping from 92% to 73% [31]. The transition from anoxic to aerobic conditions also causes hydroxylamine accumulation, which in turn fuels N_2O production via the hydroxylamine oxidation pathway. In partial nitritation–anammox systems, high aeration rates and elevated DO have been linked to increased N_2O production and enhanced gas-phase stripping due to stronger mass transfer [31]. During denitrification, oxygen plays an inhibitory role by directly suppressing the synthesis and activity of nitrous oxide reductase, the enzyme responsible for reducing N_2O to N_2. This inhibition occurs rapidly when denitrifiers shift from anoxic to aerobic conditions, while nitrite reductase activity continues, leading to the accumulation of N_2O [30].

Overall, the influence of DO on N_2O emissions is bidirectional:

- Low DO (<1 mg/L) promotes nitrite build-up and N_2O formation via nitrifier denitrification.
- Moderate to high DO (0.2–3 mg/L) shifts production towards hydroxylamine oxidation and enhances gas stripping.
- Aerobic exposure of denitrifiers inhibits N_2O reductase, causing emissions during incomplete denitrification.

This complexity highlights the importance of dynamic DO control rather than fixed setpoints. Advanced aeration strategies – balancing oxygen supply to suppress nitrite accumulation while avoiding inhibitory conditions for N_2O reductase – have been identified as effective levers to mitigate emissions in practice [28-31].

Nitrite Accumulation. Nitrite (NO_2^-) is a key intermediate in both nitrification and denitrification and plays a central role in N_2O production dynamics. Accumulation of NO_2^- is consistently linked to elevated N_2O emissions, as it can stimulate both nitrifier denitrification in ammonia-oxidizing bacteria (AOB) and incomplete heterotrophic denitrification. During nitrification, NO_2^- is formed by AOB ammonium oxidation, and elevated NO_2^- concentrations can lead directly to increased N_2O production [32, 33]. Under low DO conditions, high NO_2^- concentrations promote the expression of nitrite reductase (NiR) and nitric oxide reductase (NOR) in AOB, favouring the nitrifier denitrification pathway and thereby increasing N_2O production [34]. For example, in sequencing batch reactor (SBR) systems operated under nitritation–denitritation, elevated NO_2^- concentrations combined with DO < 1.5 mg/L were shown to significantly increase N_2O emissions [35]. This was attributed to both enhanced nitric oxide reductase gene expression and increased activity of NO_2^- reductase at higher substrate concentrations [36]. In heterotrophic denitrification, high NO_2^- concentrations have been shown to inhibit complete denitrification, resulting in the accumulation of nitric oxide (NO) and N_2O [37, 38]. Under such conditions, NiR, NOR, and N_2OR compete for electrons, and limited activity of NOR has been observed at high NO_2^- concentrations, leading to NO build-up [39]. Since NO acts as an inhibitor of denitrification enzymes, this creates a feedback loop that further favours N_2O accumulation.

Overall, nitrite accumulation acts as a potent driver of N_2O emissions in both autotrophic and heterotrophic pathways. Its impact is particularly pronounced under low DO conditions, low C/N ratios, or transient operational shifts, all of which exacerbate NO_2^- build-up. Maintaining stable and low NO_2^- levels is therefore a critical operational objective for minimizing N_2O emissions in water resources recovery facilities (WRRFs).

Rapidly Changing Process Conditions. Many studies have reported that nitrous oxide emissions increase substantially when process conditions change rapidly, such as during high ammonia loading events or under oxygen-limited conditions [83,95]. Ammonia shock loads can lead to incomplete nitrification, which in turn decreases nitrogen removal efficiency, promotes NO_2^- accumulation, and enhances N_2O production. This highlights that the overall performance stability of WWTPs strongly influences the magnitude of N_2O emissions. Oxygen limitation during nitrification is a particularly important driver, as it leads to NO_2^- build-up and N_2O formation through the nitrifier denitrification pathway. Rapid transitions in bacterial metabolism appear to require adjustment periods, during which large peaks in N_2O emissions may occur. For example, declines in dissolved oxygen concentration caused by elevated influent loading or by limitations in aeration capacity have been observed to increase N_2O production via nitrifier denitrification [40, 41]. In addition, transitions from anoxic to aerobic conditions in the presence of accumulated NH_4-N have been shown to trigger N_2O formation through the hydroxylamine oxidation pathway [42-44]. These findings indicate that abrupt changes in operational conditions – whether related to influent characteristics or aeration control – can create unstable microbial dynamics that favour N_2O production, underscoring the importance of process stability in emission mitigation strategies.

In summary, maintaining stable loading conditions and avoiding abrupt transitions in oxygen supply are key levers for minimizing N_2O emissions. This highlights the value of advanced control systems, real-time monitoring, and robust process design that can buffer or dampen sudden operational fluctuations.

Substrate Composition and COD/N Ratio. The composition of influent wastewater is a critical determinant of N_2O emissions in WRRFs, particularly with respect to the availability and type of biodegradable organic carbon. Limited availability of readily biodegradable organic carbon impedes complete denitrification, leading to the accumulation of N_2O [45], whereas excess carbon generally reduces N_2O production by ensuring sufficient electron donors for all denitrification steps. Similarly, influent nitrogen load strongly influences emissions [46], making the COD/N ratio a key control parameter in both lab-scale and full-scale systems. Several studies have highlighted the role of carbon source type in shaping N_2O dynamics. For instance, lower N_2O emissions were observed when acetate was used as the electron donor compared to methanol (resulting in EFs of 1.3% and 2.3%, respectively), being associate to more efficient N_2O reduction under acetate supplementation [47]. By contrast, it has been found that acetate addition led to higher N_2O and NO emissions than methanol, suggesting that the effect of carbon source is strongly mediated by microbial community composition and pathway diversity [48]. Similarly, when mannitol was supplied instead of acetate, N_2O conversion rates decreased significantly (21% for mannitol vs. 41% for acetate), which was linked to lower inhibition of N_2O-reductase (N_2OR) under high nitrite conditions and enhanced heterotrophic denitrification [49]. Beyond carbon type, trace metals, such as Fe(II), Fe(III), and Cu(II), can also influence N_2O dynamics by supporting or inhibiting microbial growth or by participating in abiotic reactions [50-52]. For example, insufficient Cu availability has been linked to N_2O accumulation due to impaired N_2OR activity [53]. The importance of the COD/N ratio is evident across multiple studies. Under carbon-limiting conditions, denitrification becomes incomplete because denitrification enzymes compete for electrons. Since nitrate reductase (NaR) and nitrite reductase (NiR) have a higher electron affinity than nitric oxide reductase (NOR) and nitrous oxide reductase (N_2OR), electron competition results in N_2O accumulation [39, 54]. It has also been shown that in pure cultures, 32–64% of the nitrogen load was released as N_2O under carbon-limited conditions [55]. Furthermore, microorganisms can switch to internal storage compounds as electron donors under limited external COD, further exacerbating N_2O emissions [33]. Conversely, provision of excess carbon reduces competition for electrons and supports complete denitrification, thereby lowering N_2O formation [56]. Experimental studies investigating COD/N ratios between 1.5 and 4.5 have demonstrated that the highest N_2O emissions occur at the lowest ratios [57]. Similarly, Itokawa et al. (2001) and Andalib et al. (2017) reported elevated N_2O production when COD/N ratios were below 3.5 [58, 59]. Law et al. (2012) suggested that a COD/N ratio above 4 is necessary to support complete denitrification [33], with an optimal range between 4 and 5 [60]. Gruber et al. (2021) further reported a weak positive correlation between COD/N ratio and the N_2O EF [19]. However, some studies suggest that this relationship is not universal. For example, Quan et al. (2012) found that lowering the nitrogen loading rate – or equivalently increasing the COD/N ratio – did not inhibit heterotrophic denitrification in lab-scale aerobic granular SBRs [61].

Overall, both the type of organic substrate and the COD/N ratio play decisive roles in N_2O emissions. While low COD/N ratios consistently promote higher emissions, the effect of substrate type is system-dependent, reflecting differences in microbial community composition, nitrite accumulation dynamics, and electron competition during denitrification.

pH and Temperature. pH is a major factor affecting nitrification and nitrous oxide (N_2O) emissions in wastewater treatment systems. Nitrification is highly sensitive to variations in pH [62], and biological nitrification is typically accompanied by dissolved oxygen (DO) consumption and a reduction in pH. While complete nitrification is most efficient at pH 7.5–8.0 [63, 64], the optimum pH for ammonium-oxidizing bacteria (AOB) is reported at 7.4–8.2, and for nitrite-oxidizing bacteria (NOB) at 7.2–8.0 [65]. Consequently, deviations in pH can alter the relative activities of AOB and NOB [66]. Moreover, pH influences the balance between free ammonia (FA) and free nitrous acid (FNA). High pH favors FA, which serves as a substrate for AOB but strongly inhibits NOB [67, 68]. Inhibition thresholds for FA are reported at 10–150 mg/L for AOB and 0.1–1.0 mg/L for NOB [69]. Conversely, low pH increases FNA concentrations, which inhibit both AOB and NOB, with inhibition observed at 0.22–2.8 mg/L FNA [69].

During nitrification, the highest N_2O emissions have been observed under acidic conditions. For instance, maximum N_2O production occurred at pH 6.0 [29]. NOB are particularly sensitive to acidic conditions, with no detectable activity at pH 6.5 [70], resulting in nitrite accumulation and increased N_2O generation. Interestingly, little to no inhibition has been observed at high pH values (7.5–9.95), where NOB activity remained nearly constant [70]. In partial nitrification (nitritation), accumulation of nitrites has been linked to high pH (e.g., pH 7.85 at DO = 0.7 mg/L), resulting in elevated N_2O production [71]. During heterotrophic denitrification, pH also plays a critical role. N_2O emissions were observed at pH < 6.8 [72], with maximum emissions recorded when pH decreased from 8 to 6.5, due to a reduction in N_2O reduction rates under acidic conditions [73].

Temperature is another key driver of N_2O emissions, as it affects mass transfer, chemical equilibria, and microbial growth kinetics [66]. The optimal growth temperatures for AOB and NOB are reported at 35 °C and 38 °C, respectively [74], while partial nitrification is favoured at 35–45 °C [75]. However, prolonged exposure above 40 °C leads to deactivation of nitrifiers [76].

At elevated temperatures (>25 °C), AOB growth outpaces that of NOB, which can lead to NOB washout in activated sludge systems operated at 30–35 °C. This imbalance promotes nitrite accumulation and enhances N_2O emissions via the nitrifier denitrification pathway [77].

Denitrification rates also increase with temperature, but higher temperatures reduce the solubility of N_2O, shifting emissions from the liquid to the gas phase [78]. For example, increasing the temperature from 25 °C to 35 °C reduces N_2O solubility in water by ~23% [79], directly influencing emission rates [80]. Similarly, a rise in temperature from 10 °C to 20 °C led to a 2.5-fold increase in N_2O emissions [81]. Poh et al. (2015) further showed that nitrate, nitrite, and N_2O reduction rates increased by 62%, 61%, and 41%, respectively, when temperature rose from 25 °C to 35 °C, but N_2O stripping intensified due to lower solubility, ultimately leading to higher overall emissions [78].

Finally, temperature dynamics at full-scale plants are influenced by climate zone and seasonal conditions. For instance, Gruber et al. (2021) compared N_2O emissions between WRRFs in Finland and Switzerland, attributing observed differences to influent characteristics and seasonal wastewater temperature fluctuations [19]. At the Viikinmäki WRRF in Finland, strong seasonal temperature variation was observed, with influent dropping to 8.8 °C during snowmelt in March–April, slowing reaction rates and leading to lower N_2O emissions [82].

Overall, both pH and temperature exert strong controls on microbial activity, nitrite accumulation, and solubility dynamics, making them decisive factors in determining N_2O emission levels in wastewater treatment systems.

Methane (CH_4)

Recent studies from China and India highlight significant methane contributions from the wastewater sector in these regions [83-85]. Similarly, process analyses in Mexico and Brazil demonstrate that CH_4 emissions are a major component of the sector's carbon footprint [14, 86-88]. A US monitoring study on 51 plant-wide and 33 reactor-level measurements revealed that each process unit has the potential to become a major contributor to total CH_4 emissions, though actual contributions vary widely among facilities [89]. The highest emissions were associated with sludge treatment incorporating anaerobic digestion – an order of magnitude greater than other stages such as secondary treatment – sludge treatment without anaerobic digestion, and primary treatment. Hence, methane emissions are discussed in further detail in the section “GHG Emissions from Sludge Management”.

Methane is typically produced in gravity sewers, rising mains, and pump stations, where anaerobic biofilms form under low DO conditions. Mean dissolved CH_4 concentrations of 1–6 mg/L have been reported, with higher values in warm climates and systems with long hydraulic residence times (HRT) [89]. Rising mains tend to accumulate more CH_4 per unit length but release it mainly at discharge points. Sewer-derived CH_4 can contribute up to 18% of total plant-wide emissions. Factors such as temperature, biofilm area, and wastewater composition influence CH_4 production, though data remain limited [90]. During primary treatment, biological activity is low, but stripping of dissolved CH_4 from incoming sewage or the primary clarifier can occur, particularly when CH_4-rich wastewater enters from anaerobic sewers [14, 89]. In secondary treatment, high DO levels suppress methanogenesis but promote CH_4 volatilization due to aeration turbulence. CH_4 emissions are typically concentrated at the inlet of aeration tanks and in systems with CH_4-loaded return sludge or side streams [91].

2.3.2. Indirect emissions from centralized systems

Indirect emissions from centralized wastewater treatment facilities include the following contributions:

- The use of energy (electricity and heat) from the external grid (Scope 2). The purchased energy is associated with CO_2 emissions from energy generation and transport, which are typically described as CO_2 intensity of the energy mix (g CO_2e per kWh supplied).
- The use of chemicals and materials for, e.g., external carbon dosing, phosphorus precipitation, coagulation/flocculation/dewatering processes, activated carbon treatment (Scope 3). Purchased chemicals are associated with CO_2 emissions from production and transportation to the WWTP site.
- Emissions of N_2O and CH_4 in recipients (Scope 3), resulting from the incomplete removal of COD and N in WWTPs and residual discharges with treated effluent.
- The management of sludge and biosolids (Scope 3), including post-processing (e.g., land application, incineration) and off-site transport. These emissions are detailed in the section “GHG Emissions from Sludge Management”.

Use of electricity in WWTPs is predominantly associated with aeration of biological treatment (accounting for 40–60% of the total energy use), pumping (e.g., internal mixed liquor recirculation, return activated sludge, influent pumping) and to a lesser extent mixers and other equipment. Use of heat is lower compared to electricity and is typically associated with sludge pre-heating for anaerobic digestion and heating of offices and laboratories on site.

The contribution of Scope 2 emissions over the total CO_2 footprint of a WWTP is dependent on (i) the energy efficiency of the equipment used in WWTPs (blowers, bottom/surface aerators, pumps, mixers), and (ii) the CO_2 intensity of the electricity mix at the WWTP location. In particular, the CO_2 intensity is known to have significant geographical variability, as it depends on the type of sources (fossil or renewable) used for electricity generation. Regional variability of CO_2 intensity is shown in Figure 2-3. Even within the same region, a considerable variability can occur, with EU values ranging from 8 g CO_2e/kWh (Sweden) to 594 g CO_2e/kWh (Poland) [11].

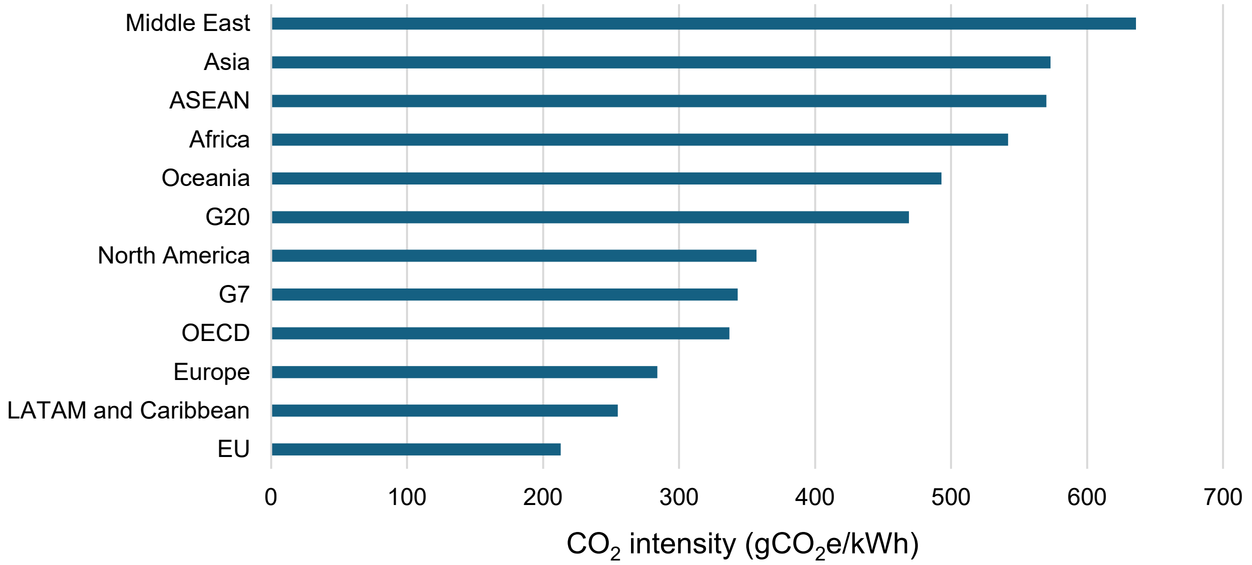

Figure 2-3. Reported CO_2 intensity of the electricity mix for 2024 in different world regions [11].

As a result, the contribution of electricity-related Scope 2 emissions over the total carbon footprint of a WWTP has been reported to vary, e.g. being <2% (Sweden, [92]), 16–28% (Denmark, [92]), from <5% to ~30% (EU, [93]) from 19 to 27% (US, [94]) and from 30% to 80% (China, [95]). It is expected that decarbonization of the energy sector, with increasing use of renewable sources for electricity generation, will contribute to the reduction of the Scope 2 emissions from WWTPs by up to 59% globally over the next 10 years [2].

The use of chemicals and other materials in treatment processes is considered to be a small (<5%), yet not negligible contributor to the CO_2 footprint of WWTPs. Typically employed chemicals exhibit a high variability in EFs, ranging (i) from 669 (methanol) to 1370 g CO_2e/kg for external carbon dosing; and (ii) from 82 (iron chloride sulfate) to 304 g CO_2e/kg (iron sulfate) for phosphorus precipitation [96]. Hence, the selection of less impactful chemicals can be crucial for reducing associated emissions. In the EU, with the implementation of the revised Urban Wastewater Treatment Directive (see section "Regional regulatory programmes"), which introduced requirements for the removal of micropollutants, such emissions are expected to increase due to the use of activated carbon treatment. The production of fresh activated carbon and its regeneration are highly resource- and energy-demanding processes, with EFs up to 3,390 g CO_2e/kg [96].

Emissions of N_2O and CH_4 are expected to occur with effluent discharge as a result of residual loads of N and biodegradable COD to recipients (and to a lesser extent, residues of dissolved

N_2O and CH_4 in effluents). In particular, N_2O emissions associated with effluent discharge have been estimated to account for up to 10% of the total CO_2 emissions from WWTPs [93].

Assessments based on life cycle assessment (LCA) methods (see also section "The role of life cycle assessment") have also estimated CO_2 emissions related to WWTP infrastructure (construction and maintenance). While being significantly lower than for sewer systems, infrastructure-related emissions for WWTPs account for 5–10% of the total CO_2 footprint [93].

Key take-aways:

- Centralized WWTPs are major and well-studied GHG sources, with N_2O often dominating Scope 1 emissions and showing extreme variability across plants and process types.
- N_2O emissions can exceed IPCC defaults by several-fold, with some WWTPs reporting >8% of influent TN emitted as N_2O, making them extreme hotspots.
- Process configuration and operating conditions strongly shapes N_2O, with BNR systems like SBRs and MBRs show the highest emissions, and operating conditions resulting in low DO and nitrite accumulation can trigger N_2O formation
- CH_4 is a significant contributor, especially from anaerobic digestion, sewers, and sludge handling, where fugitive leaks and uncontrolled anaerobic zones drive emissions.
- Scope 2 emissions depend largely on electricity carbon intensity, and global grid decarbonization will increasingly shift WWTP carbon footprints toward Scope 1 process emissions.
- Scope 3 emissions arise from chemicals, sludge management, infrastructure, and residual effluent loads, making whole-plant and life-cycle perspectives essential for accurate GHG accounting.

2.4. GHG Emissions from Decentralized Systems

Decentralized wastewater treatment systems are prevalent yet often overlooked regarding their contribution to GHG emissions. Although the emissions from a single installation may be small compared to a centralized WWTP, decentralized wastewater treatment systems serve a large portion of the global population, making their total contribution to GHG emissions significant. Decentralized wastewater treatment solutions are especially widespread in low- and middle-income countries. However, they are also a common solution in upper-middle- and high-income countries, especially where population density is low. In Japan, approximately 20% of the population is served by decentralized wastewater treatment [97], while up to one third of Irish households and 10-15% of households in Australia, USA and Canada rely on septic tank systems [98]. Decentralized systems also include small scale WWTPs, which provide at least primary treatment. For instance, most Irish WWTPs serve under 10,000 population equivalents (PE), while in Scotland around 1,600 WWTPs serve less than 500 PE [99].

2.4.1. Direct emissions from decentralized systems

Despite their prevalence, there is a lack of data on GHG emissions from decentralized wastewater treatment systems. While most research has focused on centralized systems, available data indicates that direct emissions from decentralized wastewater treatment systems are not negligible. In fact, due to their generally smaller size and less energy intensive processes, direct emissions of CH_4, N_2O and CO_2 often represent a more significant portion of their carbon footprint than for centralized systems [98, 100]. Initial findings demonstrate the need for data collection, standardized monitoring methods, and further research into the production of GHG emissions and effective mitigation strategies.

That said, studies of decentralized wastewater treatment technologies provide some insight into the process configuration and operational factors related to direct GHG emissions. Technologies where direct GHG emissions have been directly observed include:

- Septic tanks and leaching fields
- Imhoff tanks
- Soil infiltration systems (SISs)
- Waste stabilization ponds (WSPs)
- Constructed wetlands (CWs)
- Activated sludge systems, (i.e. small-scale WWTPs)

Septic and Imhoff tanks

Both septic and Imhoff tanks are designed to operate under anaerobic conditions, which mainly generate CH_4 and CO_2. Reported CH_4 and CO_2 emissions (e.g. 0.4-22 g CH_4/capita/day and 2.2-337 g CO_2/capita/day) from septic tanks are highly variable, due to temporal variations, differences in process design (e.g., number of chambers, tank seal, associated infiltration field), geographical location, and whether gas samples were collected from the top of the liquid surface or from vents [98]. Gas samples collected from vents are expected to include contributions from adjacent soil leaching fields, where tank effluent is often discharged to undergo further degradation by soil bacteria. Highly variable N_2O emissions (i.e. 0-4 g N_2O/capita/day) have also been observed [98]. There are no recent studies on emissions from Imhoff tanks.

- **Low biogas utilization.** Generated biogas is not typically utilized, since the small size of installations makes CH_4 production less reliable, while required gas cleaning (e.g. removal of H_2S and storage of combined heat and energy (CHP) is not economical at this scale. Instead, biogas streams are vented to the atmosphere.
- **Sporadic hydraulic loading** brings pronounced peaks of organic carbon, nitrogen and oxygen into the tank. Higher influent COD is correlated with higher CH_4 production, while sudden changes in nitrogen loading and DO are hypothesized to affect N_2O emissions [98].
- **Soil moisture content** affects how much air diffuses into the soil matrix, and therefore, oxygen availability. Changes in DO concentrations can promote nitrite build-up and N_2O production.

Understanding loadings dynamics and oxygen availability are critical for minimizing N_2O emissions, while direct CH_4 releases need to be reconsidered.

Waste stabilization ponds

Waste stabilization ponds (WSPs) include mechanically aerated ponds, maturation ponds, facultative ponds, and anaerobic ponds. WSPs emit high rates of CH_4 and CO_2 with high variability but can be a sink for N_2O emissions. Anaerobic ponds generate negligible N_2O emissions, while reported mean CH_4 and CO_2 emissions range from 7.0-30 g $CH_4/m^2{\cdot}d$ and 1.0-36 g $CO_2/ m^2{\cdot}d$ [15, 18, 101]. Reported mean N_2O emissions from other pond types range from -2.2 to 3.8 mg/ $m^2{\cdot}d$ [100], while CH_4 and CO_2 emissions ranged from 2.2 – 7.9 $CH_4/ m^2{\cdot}d$ and -0.7 – 129 g $CO_2/ m^2{\cdot}d$ [102, 103]. Emissions are released from the pond surface directly to the atmosphere.

- **Sludge accumulation** provides a zone of anaerobic conditions where heterotrophic denitrification can proceed, consuming produced N_2O. However, anaerobic conditions favour the formation of CH_4.
- **Organics and ammonia loading** are positively correlated with CH_4 and N_2O production. On the other hand, observed N_2O emissions decreased as influent COD concentration rose into facultative ponds [102], indicating optimal C:N ratio for N_2O consumption via heterotrophic denitrification.
- **Light exposure** affects algae's ability to produce oxygen photosynthetically in maturation and facultative ponds, as well as in high-rate algal ponds. High DO concentrations support complete nitrification, preventing nitrite accumulation.

The production and share of GHG in WSPs varies widely based on configuration type and oxygen availability as well as organics and nitrogen loading. While operator control may have limited effect, careful design and process selection are essential to minimize GHG emissions.

Soil infiltration systems

In soil infiltration systems (SIS), wastewater percolates through soil media where bacteria consume organic matter and nutrients, removing pollutants. Reported N_2O emissions are under 1.1% N_2O/ TN but may contribute up to 65% of the overall carbon footprint [100]. Since SIS operate under predominantly aerobic conditions, microbial metabolism yields more CO_2 production than CH_4. Reported mean CO_2 and CH_4 emission rates are 3.9-20 g CO_2/ $m^2 \cdot d$ and 0.09-0.68 g CH_4/ $m^2 \cdot d$ [104, 105].

- **Low C/N Ratio** contributes to compromised nitrification and incomplete denitrification, leading to N_2O production. Increasing the C:N ratio from 3:1 up to 12:1 yielded a reduction in the N_2O emission rate [106]. Similarly, diverting a portion of raw influent deeper in the soil bed can reduce N_2O emissions by providing the necessary organic carbon for complete denitrification [107].
- **Soil moisture content** affects oxygen availability in soil beds without artificial aeration. When the soil media is saturated, anaerobic conditions persist which promote denitrification but also CH_4 formation. Meanwhile, changes in DO concentrations can promote nitrite build-up, resulting in N_2O production.

Hydraulic loading is key to maintaining favourable carbon distribution and oxygen content throughout the soil media to minimize GHG emissions.

Constructed wetlands

Constructed wetlands (CWs) are a nature-based solution which combines soil, plants and microbial communities to remove pollutants. Passive aeration occurs through photosynthesis and the root systems of aquatic plants, as well as through diffusion into the soil matrix (for drained or vertical flow configurations). Reported mean GHG emission rates from CWs range from 0-2.9% N_2O/TN, 0.3-0.8 g CO_2/ $m^2 \cdot d$ and 0.1-1.3 g CH_4/ $m^2 \cdot d$ [100, 108]

- **Flow regime** influences DO concentrations in CWs, and thereby the balance of nitrifying/denitrifying (ND) processes. Surface and horizontal subsurface flow CWs are dominated by anoxic/anaerobic conditions, promoting the conversion of organics to CH_4, whereas vertical flow CWs include aerobic processes and tend to emit more N_2O due to the formation of nitrification intermediaries and limited denitrification. Similarly, GHG emissions tend to increase due to sudden changes in hydrology, for example under fill-and-drain or intermittent feeding operations [100, 108].

- **Low C/N Ratio** inhibits complete denitrification and increases N_2O formation. However, higher influent carbon content increases CH_4 and, especially, CO_2 formation. For CWs, a COD:N ratio of 5:1 is reported to avoid GHG emissions [108], but may need to be adjusted based on local conditions.

- **Plant growth** influences aerobic/anoxic conditions and thereby the microbial conversion processes that predominate. In addition, more vegetation increases oxygen transfer rate to the CW and the GHG transfer rate to the atmosphere. However, plants uptake nitrogen, reducing the amount available for N_2O formation – from the CW and CO_2 from the atmosphere – offsetting GHG emissions from microbiological processes. Specific plant species also impact GHG emissions. Research ranks *Z. latifolia* > *T. angustifolia* > *P. australis* > *T. latifolia* for highest CH_4 flux, while *Z. latifolia* > *P. australis* > *T. latifolia* > *T. angustifolia* for highest NO_2 flux [108].

Seasonal and environmental conditions impact CWs. Light exposure and temperature affect both plant and microbial growth (or dormancy). Hydrological changes influence organic, nutrient, and DO concentrations.

Decentralized activated sludge systems

As in centralized systems, N_2O flux contributes most significantly to direct emissions from activated sludge systems. For decentralized, small-scale WWTPs based on activated sludge treatment, the reported mean N_2O EFs are within the same range as EFs for large-scale centralized WWTPs (i.e., 0-2% N_2O/TN) but there is extreme variability in the data. The process configuration determines the specific operational factors with greatest influence on N_2O

emissions, as for centralized systems (see Section “GHG Emissions from Centralized Systems”).

- **Lack of sophisticated aeration controls** prevents many decentralized systems from achieving the optimal DO concentrations required to minimize nitrite build-up and N_2O formation. Intermittent aeration strategies and surface aerators can create periods or zones of intense airflow, leading to localized gas stripping. Spatial variation was significant for an oxidation system with surface aeration [109]. Initial stripping of influent CH_4 and nitrogen profile across activated sludge tanks can cause spatial variations in GHG emissions in small installations [110].

In general, the process and operational factors which influence the formation of GHG in decentralized wastewater treatment systems are also found in centralized systems. This is discussed in the section “Causes and mechanisms for Scope 1 emissions”. Additional factors unique to specific decentralized technologies may also need to be considered, including:

- Soil moisture content
- Hydraulic loading
- Light exposure

Finally, direct emissions from the transportation of treated waste (e.g. periodically emptying septic tanks, dredging ponds, etc.) offsite for disposal or further handling. Beneficial biosolids and non-potable water reuse are advantageous for agricultural areas, where population density tends to be low. Hence, decentralized wastewater treatment systems – typically located near areas that benefit from these recovered resources – offer some advantages in terms of reduced transportation distance and lower emissions.

2.4.2. Indirect emissions from decentralized systems

Regarding indirect emissions, decentralized wastewater treatment systems that rely on passive treatment (e.g., septic tanks, Imhoff tanks, waste stabilization ponds) or nature-based solutions (e.g., constructed wetlands, soil infiltration, high-rate algal ponds) consume little to no energy. Hence, their indirect CO_2 emissions from energy consumption (**Scope 2**) are insignificant. On the other hand, small-scale WWTPs serving a small, local population (i.e. <1,000–10,000 PE) typically employ the same treatment processes as centralized, large-scale WWTPs. As these WWTPs generally do not have advanced process controls and energy-efficient equipment, their Scope 2 emissions may make up a significant portion of their carbon footprint.

Small-scale WWTPs may also require chemical inputs, whereas passive treatment systems generally do not. The indirect emissions associated with chemical consumption fall under

Scope 3, along with those from construction. Unlike chemical consumption, which represents an ongoing impact, construction-related emissions occur mainly during the building phase. The type of technology governs the available construction methods and materials, so options should be compared. For instance, a comparison of decentralized treatment options for a municipality in the Ladakh region of India found that individual household latrines would have considerably higher Scope 3 emissions (26,365 t CO_2e) than either a small-scale WWTP (2,177 t CO_2e) or nature-based decentralized wastewater treatment systems (DEWATS) (1,153 t CO_2e), mainly due to excavation and concrete materials [24]. Where appropriate, selecting concrete alternatives, such as high-density polyethylene (HDPE) tanks, can have a lower life cycle environmental impact in terms of GHG emissions and the depletion of common construction materials [98].

In terms of off-site emissions, a decentralized wastewater collection system typically has lower pumping energy requirements and less of a construction impact compared to a centralized one. Furthermore, fewer transportation losses mean it is more efficient to provide the infrastructure for district heating from wastewater heat recovery or biogas for fuel from these systems locally. However, where sewage is removed and trucked treatment locations, off-site fuel- and energy emissions may be considerable.

Key take-aways:

- Decentralized systems serve a large share of the global population, making their cumulative GHG emissions considerable despite small per-system footprints.
- Direct CH_4 and N_2O emissions are often proportionally higher than in centralized plants because processes are simpler, uncontrolled, and rarely monitored.
- Septic tanks, Imhoff tanks, ponds, wetlands, soil-based systems, and small WWTPs all emit GHGs, with emissions heavily shaped by oxygen availability, hydraulic loading, and C/N ratios.
- Septic tanks and ponds are major CH_4 sources, while soil systems and wetlands can generate notable N_2O under low-carbon or fluctuating moisture/oxygen conditions.
- Owing to reduced energy and construction material requirements, nature-based systems are characterized by comparably small Scope 2 and Scope 3 emissions.

2.5. GHG Emissions from Sludge Management

Sewage sludge is a byproduct of both centralized and decentralized treatment systems. As small decentralized systems typically do not have the capacity to manage sludge (e.g., septic sludge), sludge management often takes place in centralized facilities. Sludge management includes the following steps:

- Sludge processing (e.g., dewatering to reduce volumes)
- Biological sludge stabilization, most commonly through anaerobic digestion
- Post-processing of stabilized sludge through post-dewatering and/or dedicated processes (composting, incineration)
- Final disposal (land application or other use destination)
- Treatment of concentrated reject water generated from dewatering processes

GHG emissions occur at all steps in the sludge management cycle (Figure 2-1):

> **Scope 1** emissions including fugitive CH_4 and N_2O emissions from sludge storage/dewatering/digestion and reject water treatment
>
> **Scope 2** emissions from energy used for sludge on-site transport (i.e. pumping) and from heat used for sludge pre-heating for high-temperature processes (e.g., anaerobic digestion)
>
> **Scope 3** emissions from the use of chemicals to improve sludge dewatering and stabilization, as well as emissions associated with off-site sludge transport and disposal

Sludge processing and stabilization

Methane (CH_4) emissions are the main GHG emissions in sludge processing and stabilization processes. As sewage sludge is rich in organic matter, and anaerobic conditions are prevalent at various treatment stages, methane production is facilitated.

Processing steps typically include sludge storage and dewatering (e.g., through thickening processes), in which methane emissions occur as a result of open-air systems and/or leaks in process units due to poor sealing. Anaerobic digestion is a commonly used biological sludge stabilization technology which converts the organic matter in sludge to methane-containing biogas. In anaerobic digesters, fugitive losses can occur from leaks in reactors and gas pipes, including biogas conveyance to combined heat and power systems (see section “GHG Emissions from Energy and Resource Recovery Practices”). Anaerobic digestion is typically combined with pre- and post-dewatering steps, and temporary storage of both undigested and digested sludge.

In WWTPs with anaerobic digestion, it has been estimated that methane emissions from sludge processing contribute up to 72% of total CH_4 emissions from a WWTP, with 50% from storage units and 10% from dewatering [111]. A country-wide survey in Denmark showed that fugitive methane emissions from leaks in WWTPs accounted, on average, for 6.7% of total biogas production, reaching up to 40% [112]. Using updated datasets, plant-wide CH_4 emissions in United States were estimated to be 10.9 Mt CO_2e/y, approximately twice as high as previously estimated using Tier 2 methods, with most emissions originating from WWTPs with anaerobic digestion and stabilization ponds [89]. Recent evidence, gathered through drone-based measurements for selected WWTPs in Sweden, confirmed this finding, estimating methane emissions from sludge processing (including anaerobic digestion and storage) to be 2.5 times higher than originally estimated using IPCC EFs [113]. On the other hand, experience from Mexico indicates that default Tier 1 EFs highly overestimate CH_4 emissions compared to on-site measurements [88]. Therefore, the importance of on-site measurements is reiterated for implementing local mitigation plans as opposed to preparing high-level, national estimates.

Traditionally neglected, nitrous oxide emissions from sludge processing have also been found to be considerable, occurring mainly during storage and dewatering stages as part of unintended aeration of sludge [113, 114]. While only 9% of CH_4 emissions on mass basis, at scale N_2O emissions were found to be equivalent to CH_4 emissions on CO_2e basis, owing to their larger global warming potential [113]. Overall, these findings highlight the global need for:

- long-term monitoring and improved quantification methods to better estimate fugitive CH_4 emissions
- the inclusion of N_2O emissions from sludge processing steps and improvements in standard calculation methodologies.

Post-processed biosolids handling

While generally located outside a WWTP, additional processing of stabilized biosolids can result in additional GHG emissions. Options include:

- Incineration
- Composting
- Landfilling
- Pyrolysis / gasification
- Application on land as fertilizer

It should be noted that these processes are also used (e.g., after chemical sludge stabilization) to replace sludge processing.

While limited experimental reports from full-scale systems are available, attempts have been made to estimate GHG emissions from biosolids' post-processing and destination. A number of studies have compared different processing options, showing overall that landfilling had the one with highest GHG emissions followed by incineration, gasification, composting and land application [115-117]. Nevertheless, uncertainties in calculation methods still persist (resulting from, e.g., local conditions), and other studies have associated higher emissions with composting than with incineration [118].

Due to transportation to agricultural fields and N_2O and CH_4 emissions in soil, land application can still be a substantial source of GHG emissions (while contributing to savings in fertilizer use) [119]. It has been estimated that land application can account for up to 20% of the total CO_2 footprint of a WWTP, with an additional 10% contribution from temporary biosolids storage (stockpiling) prior to land application [92].

Treatment of reject water

In addition to mainstream bioreactors, sidestream processes for treating high-strength wastewater generated from sludge dewatering processes represent a potentially significant source of N_2O emissions. Reported data indicate that the mean EFs in sidestream processes can be more than twice as high as in mainstream processes [120]. This increase can be attributed to high ammonia load, with side-stream processes typically receiving streams at 500–1,500 mg N/L levels, compared to 20–70 mg N/L in influent wastewater, creating conditions that facilitate N_2O production [64].

Wide variability exists among sidestream technologies [64, 121]:

- Nitrification-denitrification (ND) processes can exhibit up to seven times higher mean EFs compared to partial nitritation-anammox (PNA) processes
- One-stage PNA systems exhibit significantly lower emissions compared to two-stage systems, as one-stage PNA limits nitrite accumulation, which is usually correlated with greater N_2O emissions
- Attached-growth PNA systems (e.g., using moving bed biofilm reactors) exhibit considerably (7- to 8-fold) lower EFs compared to suspended growth PNA systems

These findings suggest that while sidestream processes have higher emission potential, technology selection and operational optimization can substantially reduce N_2O emissions.

2.6. GHG Emissions from Energy and Resource Recovery Practices

Wastewater treatment, in particular sludge management, can be important sources of renewable resources, including energy (power and heat), biofuels, carbon, nutrients and water itself. Commonly used energy and resource recovery technologies include:

- Biogas production and recovery of energy (through combined heat and power systems) or biogas upgrading (for biomethane production)
- Heat pumps for heat recovery from wastewater
- Recovery of phosphorus for use as fertilizer
- Recovery of added value organics, e.g., biopolymers and biochar

These approaches can generally contribute to reducing GHG emissions by (i) providing alternative energy and heating sources, making WWTPs net energy suppliers instead of consumers and therefore displacing associated Scope 2 GHG emissions from WWTPs; and (ii) supporting agricultural and industrial sectors through recovered resources, offsetting their GHG emissions related to the sourcing and transport of water, fertilizers, and polymers, thus effectively contributing to the circular economy and the food-water-energy nexus. Therefore, the benefits associated with these approaches will be discussed in more detail in the section "GHG Emissions Mitigation Strategies from Wastewater Treatment Systems".

Nevertheless, technologies used for energy and resource recovery can also be associated with GHG emissions, e.g.:

- **Biogas upgrading and biomethane production:** Biogas is processed to higher purity standards with a minimum 90% methane composition. The process of upgrading biogas to biomethane requires the removal of water vapour, CO_2, H_2S and other impurities. Then, gas compressors pressurize the conditioned gas to its final form, either compressed natural gas (CNG) or liquefied natural gas (LNG). The water vapour condensers, membrane purification systems and gas compressors are energy-intensive equipment, exerting indirect CO_2 emissions. Furthermore, the activated carbon and polymeric membranes used to remove H_2S and impurities, are derived from coal and fossil-based synthetic polymers. Hence, these materials also contribute indirectly to the GHG emissions. Finally, leaks or releases during the storage, upgrading and distribution of biogas and biomethane contribute to direct emissions.
- **Combined heat and power (CHP) facilities:** CHP, also known as a co-generation system, generates multiple forms of energy (i.e. electricity and heat/steam) from biogas. CHP systems directly produce emissions, including CO_2, from the combustion of biogas, and CH_4 and N_2O from leaks or residual flue gas.
- **Recovery of phosphorus from solid matrices:** Wet-chemical and thermochemical methods are applied to recover phosphorus from dewatered or dried sewage sludge or

incinerated sludge ash. Up to 98% of phosphorus can be recovered from dried sludge or sludge ash [99, 122]. The wet-chemical approach is more widespread but is associated with chemical use (e.g. sulfuric or hydrochloric acids), use and regeneration of organic solvents, and the generation of liquid waste containing heavy metals, which requires additional treatment. The thermochemical approach requires less chemical input and does not produce liquid waste, resulting in potentially lower Scope 3 emissions. Yet, both approaches consume energy, in the range of 0.33 – 26.5 kWh/kg P recovered [122], contributing to Scope 2 emissions. Thermochemical processes also often require a fuel source (e.g. natural gas) initially; however, there is the potential to recover heat once the process is under way.

- **Recovery of phosphorus from liquid matrices:** Chemical precipitation, such as struvite or calcium phosphate precipitation, is a well-established technique from P recovery, especially from concentrated process streams from sludge dewatering. Precipitation requires significant chemical additions (Scope 3) for pH adjustments, first to increase the pH for alkaline precipitation conditions, then to reacidify the process effluent. Additional chemicals are used as flocculants (polymers) and to maintain the equipment and piping associated with precipitation processes, which are often subject to scaling and clogging. Alternative phosphorus recovery technologies include the application of adsorbents and are well-suited for decentralized installations. Certain adsorbent materials (e.g. crab carapace micropowder CCM) can filter phosphorus from the water, though this technology is most effective on wastewater with low turbidity (e.g. secondary effluent or filtrate) [123]. The thermochemical activation and optional regeneration of the adsorbent contribute to both Scope 2 and 3 emissions. Instead of regeneration, bio-adsorptive materials could potentially be used as a soil amendment.
- **Nutrient recovery using algae:** Algae-based processes have shown potential in harvesting nitrogen and phosphorus from wastewater. Depending on the type of system employed, algae treatment can have variable energy consumption to meet light intensity and temperature needs as well as pumping, filtration, and drying of harvested algae. Furthermore, some systems may require chemical additions to maintain a fixed pH for optimal growth. Specific algae monocultures have even shown promise to replace secondary treatment at small-scale WWTPs [124], thereby eliminating energy-intensive aerobic treatment and its associated emissions.
- **Recovery of added value organic products:** Processes to recover biopolymers from sludge require heating and pH adjustments as well as separation and purification techniques. GHG emissions' accounting should thus consider additional power and chemical consumption (Scope 2 and 3). Scaling these technologies to industrial and commercial use is ongoing, so the relevant climate impact still needs to be determined.

Key take-aways:

- Sludge management generates significant GHGs across all stages—processing, stabilization, post-treatment, disposal, and reject water treatment—with CH_4 and N_2O as major contributors.
- Anaerobic digestion is a key source of CH_4 emissions, often underestimated by standard emission factors, highlighting the need for site-specific monitoring.
- Post-processing and disposal of biosolids can result in additional GHGs, with landfilling representing the least beneficial alternative.
- Energy and resource recovery represent attractive and technologically-mature options to reduce GHGs by displacing the use of fossil-derived energy sources and products

3. Monitoring of GHG emissions

This chapter describes the drivers, objectives and methodologies used for monitoring GHG emissions in wastewater treatment systems, with specific focus on Scope 1 emissions.

3.1. Applications of emission monitoring

The monitoring of Scope 1 GHG emissions from WWTPs mainly quantifies direct emissions of CH_4 and N_2O – the most potent gases emitted from wastewater treatment. Direct CO_2 emissions can also be monitored using similar techniques. However, most CO_2 emissions from the wastewater sector come from power consumption and indirect sources (Scope 2 and 3).

Policymakers and government agencies may call for monitoring of direct emissions from WWTPs to update national or regional EFs and to establish benchmarks for the wastewater sector. In 2020, the Danish EPA adopted a new national EF for N_2O after an extensive measurement campaign, which found the calculated EF (i.e. 0.84% kg N_2O-N/kg TN) was about half the value in the 2019 IPCC report but 2.5 times greater than the previous national EF [45]. Meanwhile, an extensive monitoring study of over 63 WWTPs in the USA concluded that actual CH_4 emissions are likely 1.9 times greater than the current US EPA's inventory, based on IPCC guidelines [125]. However, many nations rely solely on the IPCC's Tier 1 EFs, having not yet established national EFs which would be more accurate for local practices and regional conditions.

Wastewater utilities, plant managers and operators may use emissions monitoring to determine specific EFs for their local WWTP. Recent studies in China, the USA and the Netherlands indicate that general EFs are not representative for many WWTPs [26, 125-127]. The Dutch Foundation for Applied Water Research (STOWA) concluded that the high variability of N_2O over time and between WWTPs makes a single EF impossible, and instead recommended risk evaluation and local monitoring of high risk WWTPs based on influent NH_4 and NO_2 concentrations [126]. Thus, local emissions monitoring is essential to identify emissions hotspots, to understand site-specific triggering processes and operational conditions for CH_4 and N_2O formation, and to develop and track mitigation strategies for reducing emissions.

3.2. Monitoring approach and campaign period

Depending on the goals of the emissions monitoring campaign, the appropriate monitoring approach and analysis method may vary. The first decision to be made is whether the monitoring campaign will measure plant-wide emissions or focus on specific process units.

The plant-wide approach provides data on overall emissions from the site, including any unknown leaks as well as process units that may be difficult to measure individually. Ground-based remote sensing methods are used to monitor emissions from a distance downwind of the WWTP to capture total site emissions. Thus, monitoring does not interfere with daily operations. For remote sensing systems, weather patterns, road conditions and accessibility to adjacent sites also need to be considered. The plant-wide approach is well-suited to provide overall emissions estimates and calculate site EFs. It can also be used to identify main emission sources ("hotspots") by deploying remote sensing equipment on site to capture overall emissions from specific process areas [22].

On the other hand, monitoring specific process units is necessary to improve our understanding of the formation and release of CH_4 and N_2O. Focused monitoring facilitates the correlation of emissions patterns with the multiple process and operational parameters at play in a single unit, which are essential for identifying triggers and to develop a site-specific mitigation plan. This approach can also support a bottom-up approach to estimating total WWTP emissions, when the hotspots are individually monitored and summed. Process unit monitoring relies on either grab sampling and analysis or continuous measurements. Hence, monitoring results are specific to the location of sample collection and are not necessarily representative of the entire process unit or specific zones of the process unit.

Deciding appropriate monitoring periods and sampling frequency is critical to ensure that the campaign is representative of typical WWTP emissions. A short monitoring period may overlook significant seasonal variations in emissions, as temperature and operational changes are known to affect the formation of GHG. Studies that investigated for seasonal trends reveal high variability N_2O emissions and tend to report higher EF (median 1.7% kg N_2O-N/kg TN) than short-term studies (median 0.2%), indicating that short-term campaigns may underestimate or yield unreliable EFs [27]. There is a similar discrepancy in reported EFs between short-term (0.18% kg N_2O-N/kgTN) and long-term (1.41%) N_2O monitoring campaigns [26].

Hence, the monitoring period must cover the full scale of temporal variations. Although resources for and access to monitoring equipment may not be available for full-year or multiple months, an appropriate monitoring period can be achieved by including collection days representative of seasonal and operational variations. The occurrence and timing of seasonal effects varies by region, while operational changes are specific to each WWTP

Similarly, sampling frequency affects the interpretation of results. Online monitoring can reveal diurnal and short-term emissions patterns, which grab samples may miss. In fact, studies that implemented continuous monitoring of N_2O emissions at WWTPs reported a higher EF (median 1.1% kg N_2O-N/kg TN) than studies that used discontinuous sampling (median EF was 0.2%) [27].

Yet, grab samples monitoring campaigns can also provide useful information, if a discreet sampling frequency that accounts for short-term variations is adopted. At a minimum, sampling frequency should consider known daily maximum and minimum loading to the plant. In addition, variability between weekdays and weekend operations, such as sludge handling and the return of process or sludge dewatering flows, may impact emissions and should be included when planning a sampling programme.

3.3. Monitoring approaches: Sampling and analysis methods

Once the monitoring approach and campaign period are decided, the appropriate sampling and analysis methods can be selected. The analysis method dictates the type of sample required, and therefore, the proper sampling method. Several analysis methods are available to detect and quantify CH_4 and N_2O concentrations. Selection of the appropriate option depends on local availability, site accessibility, cost, and calibration and maintenance needs (typically performed by trained operators).

3.3.1. Sampling methods

The sampling method must be compatible with the selected analysis method and sample phase (e.g. dissolved in liquid or gaseous).

Liquid samples must be collected under the water surface to prevent atmospheric interferences on the dissolved gas concentrations. Grab samples must be withdrawn from below the surface using a syringe to transfer to a sample vial while excluding air. The headspace method can be used to extract dissolved gases from the liquid phase for recovery and analyses using gas chromatography (GC) to determine maximum emissions potential using Henry's law (e.g. ASTM D8028-17). For continuous measurements, N_2O sensors located in the tank below the water surface can be used. Commercial CH_4 sensors are not commonly used in wastewater settings, since they are primarily designed for clean water application and are highly sensitive to impurities and sulfide concentrations [120]. Importantly, liquid grab samples and sensors are representative of the specific sample location but not necessarily of the entire process unit.

Gas samples can be collected from ventilation systems of covered process units or from the off-gas of process units open to the atmosphere (e.g. uncovered oxidation ditches). In either case, gas can be collected as grab samples and stored in sample bags or tubes for offline analysis (i.e. GC). Alternatively, a portion of the gas flow can be diverted to an online gas analyser for continuous monitoring.

A floating hood space typically is used to isolate and collect off-gas for sampling and analysis. This collection method requires a gas flow – either from off-gas above an aerated unit or an induced gas flux over unaerated units. For unaerated units, it is recommended that the gas flux matches the windspeed [25]. While WERF protocol certified by the US EPA uses SEIFC hoods [120], floating hoods can come in different configurations and simplified adaptations to reduce instrumentation requirements [128]. To overcome gas building up under the hood, which would change the partial pressure as the sample collection proceeds and would reduce the measured off-gas rate after several minutes, hood configurations are ventilated or use a sweep gas, allowing for prolonged collection times. Importantly, this type of sample collection is indicative of off-gas conditions for the area under the hood, so multiple samples are recommended from at least 2 locations and up to 2% of the tank surface area per ASCE 18-96 standard for off-gas measurements [25]. Fortunately, floating hoods can be easily moved to collect measurements from multiple positions and to compare emissions across the process unit, i.e. anoxic versus aerated zones of an activated sludge basin.

For fully covered facilities, gas samples can be collected from the ventilation system, allowing calculation of total emissions from a WWTP [129]. Although it cannot be used to pinpoint where in the process emissions originate, this method provides for a reliable way of determining total direct emissions, their temporal dynamics, and deriving WWTP-specific EFs.

3.3.2. Analysis methods

The main analysis methods for measuring CH_4 and N_2O concentrations are summarized below.

Gas chromatography (GC) analyzes a gas sample offline in a laboratory setting, due to instrument sensitivity and (often) pre-analysis sample preparation. Dissolved gases can also be analysed using the headspace method, where a chemical reaction drives dissolved gas to form the liquid sample; gas collected in the headspace is then analysed (e.g. ASTM D8028-17, US EPA 5021A). In GC analysis, a chromatographic column separates the components, so that each gas enters the detector at a specific elution time. The detector measures a response, which is calibrated to represent a concentration for each component gas. For CH_4, a flame ionization detector (FID) is paired with the GC, while for N_2O an electron capture detector (ECD) is used. Due to the high cost of a GC analysis system and the need for trained instrument operators to calibrate and maintain the instrument, run the analysis, and interpret results, the WWTP typically sends collected samples to commercial or university laboratories to perform this analysis. Sampling costs typically range from € 30 – 200 per sample, depending on the sample preparation required and local laboratory pricing.

Infrared (IR) spectroscopy detects gas compounds by passing IR radiation through the sample and measuring the absorptions made by bonds at specific frequencies – the gas' "fingerprint" so to say. Both non-dispersive IR and Fourier transform IR spectroscopy methods are used to detect CH_4 and N_2O gases. The IR spectrometer should be periodically calibrated to relate absorption response to gas concentration. To detect emissions, the target gas must be distinguishable from background concentrations. Practical IR spectroscopy devices used in the field are the continuous gas analyser and gas imaging camera.

- A continuous gas analyser (e.g., Emerson-Rosemont) measures concentrations in the gas flow entering the device. Gas analysers can be mounted to a ventilation or off-gas system, where gas flow is routed directly to the device. Since they can also detect CH_4 and N_2O gases in ambient air, gas analysers can be mounted to a stationary location or mounted onto a vehicle or drone to take mobile measurements across the WWTP, as a ground-based remote sensing method. Reported costs from a continuous gas analyzer are € 10,000 – 25,000 [126]. Meanwhile, remote gas analysis using a drone mount is estimated to be € 5,000 based on experience in Denmark [130].
- Alternatively, emissions can be detected remotely using an IR imaging camera with an absorption filter for a specific wavelength (e.g., GasFindIR camera FLIR GF320). Imaging cameras typically monitor an entire process unit (e.g. leak detection) or a specific portion of the process area (e.g. ventilation outlet). Using a similar concept, a hyperspectral thermal imaging camera can detect gases by absorption of thermal radiation, where each image pixel provides a spectrum. This new technique can capture images at a high frequency (i.e. hundreds per second), the data can also be analysed to interpret gas velocity and dispersion, assisting the calculation of emissions [131]. IR imaging cameras range from € 2,000 – 30,000 while imaging software is on the order of € 2,000 – 5,000.

Amperometric sensors measure the dissolved N_2O concentration in the liquid phase based on an electrochemical reaction between N_2O molecules and a cathode in the sensor. The sensor provides continuous measurement, which can be paired to SCADA. Several studies have shown excellent correlation between the measurements from N_2O liquid sensors and gas analyser measurements, yet emissions estimated from the liquid sensor may differ slightly (for instance, on average 22% higher in Amsterdam West) than those from a continuous gas analyser [126]. Gas emission estimates based on liquid phase concentrations need to be converted using a mass transfer coefficient (kLa) for N_2O, which varies based on airflow, temperature, aeration configuration and reactor dimensions. The standard conversion model is currently only valid for bottom-aerated (i.e. bubble aeration) reactors [121]. Surface aeration configurations require site-specific calculations with spatial modelling of the kLa across the tank [109]. The dissolved N_2O sensor (e.g. Unisense Environment) requires calibration – at least every 2 months or when temperature changes more than 3 °C – as well as regular replacement of the sensor. The cost of an amperometric sensor ranges from € 15,000 – 20,000 plus periodic replacement of the sensor membrane (about € 1,000) [126].

Eddy covariance (EC) is a traditional method used to measure the turbulent fluxes of temperature and trace gases between the land surface and the atmosphere. EC has recently been applied to measure N_2O fluxes from aeration tanks based on the principle of turbulent transport in the atmospheric surface layer. It calculates surface gas fluxes from the covariance between vertical wind speed – measured with a three-dimensional sonic anemometer – and gas concentrations recorded by a high-frequency gas analyzer [132]. The resulting fluxes are attributed to an upwind source area defined using a flux footprint model, the extent of which depends on factors such as wind direction, measurement height, surface characteristics, and atmospheric stability. The advantages of the EC method include its ability to provide continuous, automated, long-term measurements that capture both short-term and seasonal variations. In addition to the cost of the gas analyzer, there is the cost of the sonic anemometer for wind speed measurements (cost range € 10,000- 15,000) and software for analyzing the data and calculating fluxes.

Table 3-1: Overview of analysis and sampling methods for monitoring CH_4 and N_2O emissions.

Method	Targets (phase)	Analysis	Sampling	Setting
Gas chromatography (GC)	CH_4 (gas) N_2O (gas)	Offline, ex situ	Grab samples	Process unit
Infrared (IR) gas analyzer	CH_4 (gas) N_2O (gas)	Online, in situ Remote Offline, ex situ	Continuous Continuous while deployed Grab samples	Plant-wide Process unit (covered)
IR or thermal imaging camera	CH_4 (gas) N_2O (gas)	Remote	Continuous while deployed	Process unit
Amperometric sensor	N_2O (liquid)	Online, in situ	Continuous	Process unit

3.3.3. Calculating emission rates and factors

The analysis of the gas or liquid samples yields a concentration, which then needs to be converted to an **emissions rate** (e.g. kg per day, tons per year) and an **emission factor (EF)** for the monitoring data to be meaningful and comparable. Direct measurements of CH_4 and N_2O concentrations are coupled with gas flows to calculate the emissions rate, so the measured gas flow must be reliable. Additionally, dissolved gas concentrations require an accurate mass transfer coefficient from the liquid to the gas phase to estimate emissions. The mass transfer coefficient can be determined theoretically, empirically or by oxygen proximity. The latter relies on continuous dissolved oxygen (DO) measurement, and is ideal for capturing the dynamic mass transfer conditions found at most WWTPs, where the aeration rate changes throughout the day.

Remote sensing methods (i.e. gas analysers or gas imaging cameras) monitor emissions in the ambient air, meaning several gas flows may be captured from different units and at different rates as the gases disperse in the atmosphere. Hence, remote sensing methods calculate emissions by describing a gas plume and defining atmospheric dispersion by downwind gas concentration measurements. While various atmospheric dispersion models have been used to track CH_4 emissions in biogas and landfill industries, the mobile tracer gas dispersion method (MTDM) is the only method used in WWTPs to monitor both N_2O and CH_4 emissions [120]. MTDM uses a tracer gas, released at a known concentration and flow rate, and simultaneously measures tracer and target gas concentrations downwind from the source, traversing the tracer plume. The dispersion of the target gas is approximated from the observed tracer gas plume, from which the emissions rate is determined [22]. The MTDM method has been widely applied in Scandinavia, and is a recognized monitoring method by the Danish EPA.

Once the emissions rate is established, **the calculated emissions factor (EF)** relates the emissions rate to the influent COD or nitrogen loading and removal (influent minus effluent load) at the plant. CH_4 and N_2O emissions can also be converted to CO_2 equivalent to facilitate determination of the overall carbon footprint of the WWTP and ranking of processes with the largest impact to prioritize mitigations.

3.3.4. Recommendations for emission monitoring

While emissions monitoring is the most accurate way to determine direct emissions from WWTPs, the availability of resources (i.e. costs, man/woman-hours, equipment) challenges widespread implementation of monitoring programmes. In the future, advances in the field of gas emissions measurement will hopefully reduce costs and simplify detection so that direct emissions monitoring will become more common place. Given current constraints, the following guidelines are provided:

1. **Define the goals of the direct emissions monitoring programme.** A plant-wide monitoring approach may be sufficient for estimating EFs and emissions benchmarking, but process unit monitoring is needed to understand GHG formation mechanisms and to develop a mitigation plan. Monitoring goals as well as local requirements affect the selection of appropriate methods for detecting and quantifying CH_4 and N_2O.

2. **Prioritize where to implement the programme.** Direct emissions monitoring has the greatest impact, where N_2O or CH_4 emissions contribute significantly to a plant's carbon footprint. There are certain processes associated with the generation of GHG (e.g. anaerobic digestion for CH_4, biological nitrogen removal in activated sludge treatment basins for N_2O). Moreover, experience in the Netherlands has linked effluent ammonia and nitrite concentration to N_2O emissions and thereby created a method to assign emissions risk to WWTPs [126]. By reviewing process diagrams and effluent

concentration data, WWTPs with a higher risk of direct emissions can be prioritized for monitoring.

3. **Preform a preliminary assessment.** A preliminary assessment provides an order of magnitude analysis and confirms whether high-risk WWTPs or expected hotspots are indeed major sources of direct emissions. Ideally, long-term, continuous monitoring would provide the most complete emissions data, but intelligent choices in sampling frequency and monitoring periods can provide representative measurements.
4. **Verify hotspots.** Monitoring known hotspots and summing process unit emissions (bottom-up approach) is an acceptable method to estimate overall WWTP site emissions, where plant-wide monitoring (e.g. remote sensing) cannot be implemented.
5. **Record plant data.** Dimensional (e.g. tank surface area), operational (e.g. airflow) and performance (e.g. influent/effluent concentrations) data are required to calculate emission rates and EFs from measured gas concentrations. These data are also essential to verify that the sampling campaign fully represents temporal variations, to identify process parameters or performance indicators that are indicative of GHG production, and to calibrate process models.
6. **Use process models as an estimation tool.** There are several mechanistic wastewater process models available, which can simulate the dynamic formation and emission of N_2O and CH_4, instead of relying on fixed, generic EFs. Process models can be a useful tool to estimate direct emissions between monitoring periods or when a monitoring programme cannot be implemented.

Case #1: Geographical variability in reported GHG emissions from WWTPs

Country/region: EU (Denmark, Sweden), USA, China, Australia

Background: While clear guidelines are available for the calculation of GHG emissions from WWTPs, limited publicly available data exist at national and regional level. Limitations include (i) the combined calculation for collection systems and WWTPs, (ii) the difference between default (Tier 1) and country-specific (Tier 2) EFs, and (iii) modifications to parameters used in calculations (e.g., global warming potentials for N_2O and CH_4).

PE-normalized total emissions

Europe: Parravicini et al. [93] has estimated emissions from operational WWTPs to be 20.5 Mt CO_2e/y, corresponding to 34 kg CO_2e/PE/y (25–75 kg CO_2e/PE/y, depending on the type and size of WWTP). Earlier studies have estimated country-specific emissions to be 77 kg CO_2/PE/y for Greece [133] and, based on studies on selected WWTPs, 40–200 kg CO_2/PE/y for Italy [134] and 11–48 kg CO_2/PE/y for Denmark and Sweden [92].

USA: A recent study [94] has estimated country-wide emissions from WWTPs to be 47 Mt CO_2e/y (41–55), corresponding to 95 kg CO_2e/PE/y (83–111).

China: owing to the increase of the number of WWTPs entering in operation in the last two decades, a corresponding increase in the CO2 emissions from the wastewater treatment sector has been reported [135]. Most recent calculations for the year 2020 have estimated emissions from Chinese WWTPs to range between 31 [135] and 56 [95] MtCO_2e/y, corresponding to 41–75 kg CO_2e/PE/y.

Figure 3-1 provides a preliminary comparison of estimated PE-normalized EFs for total CO_2 emissions from WWTPs. *Considering the underlying uncertainties affecting the estimates, this comparison is to be regarded as indicative.*

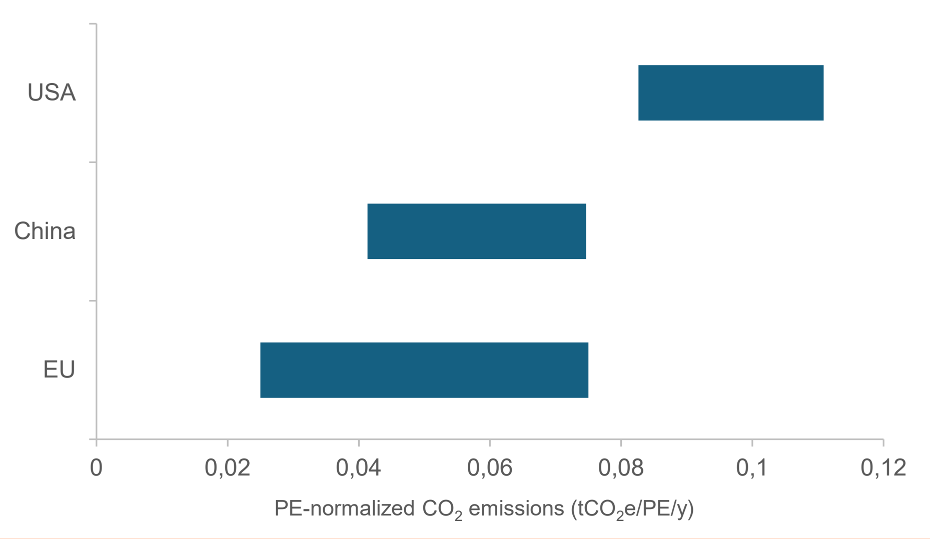

Figure 3-1. Comparison of PE-normalized CO_2 emissions from WWTPs in EU, China and USA.

N_2O emission factors

Monitoring data collected from a number of countries have allowed calculated country-specific EFs for N_2O in WWTPs [120] [64, 121]. While considerable variability was shown within each country (0 to 6% kgN_2O-N / kgN), clear differences were shown:

- ***Australia:*** median EF = 1.35% kgN_2O-N / kgN, mean EF = 1.6% kgN_2O-N / kgN
- ***China:*** median EF = 0.2% kgN_2O-N / kgN, mean EF = 0.8% kgN_2O-N / kgN
- ***Denmark:*** mean (weighed) EF = 0.84% kgN_2O-N / kgN
- ***Sweden:*** median EF = 0.74% kgN_2O-N / kgN, mean EF = 0.9% kgN_2O-N / kgN
- ***USA:*** median EF = 0.3% kgN_2O-N / kgN, mean EF = 0.4% kgN_2O-N / kgN

4. Policy instruments for GHG emissions reduction

4.1. Closing the gap on untreated wastewater (SDG 6.3)

Sustainable Development Goal 6.3 states: "*By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally*". Indicator 6.3.1 tracks the proportion of wastewater generated by domestic and industrial sources treated safely.

While it had been estimated that 80% of wastewater generated worldwide is untreated, this estimate has recently been updated to a still considerable 44% of household wastewater not being safely treated [136]. The new estimate is based on partial data, with major gaps in highly populated regions such as South America, Africa and South Asia (Figure 4-1).

The fulfilment of SDG target 6.3, while contributing to improved sanitation, public health and water quality, is also expected to increase overall emissions from the wastewater treatment sector. It has been estimated that new WWTPs required to achieve this target will lead to an increase in CO_2 emissions from the wastewater sector by 0.29–0.39 Mt CO_2e/y [1]. Thus, knowledge-informed planning as well as targeted policy instruments are required to minimize the emissions' impact from new WWTPs.

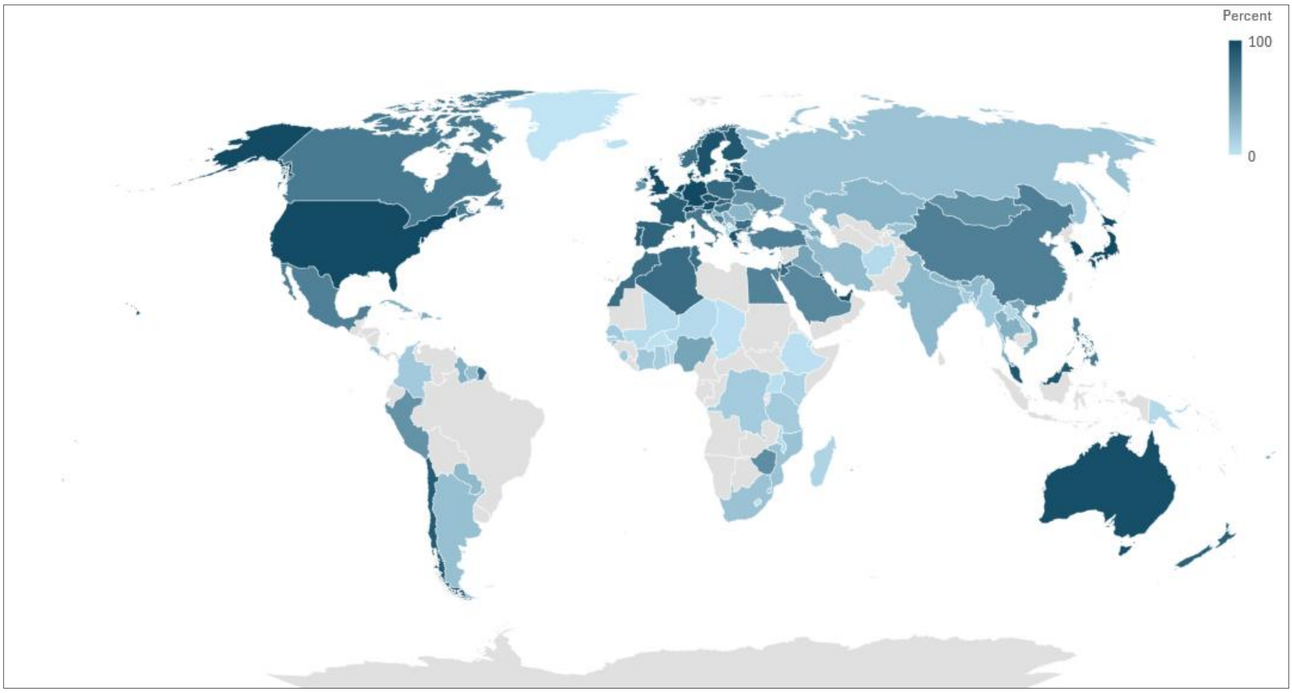

Figure 4-1. Portion of safely treated domestic wastewater flows from WHO's 2025 country files for SDG 6.3.1 [136].

4.2. Examples of regulatory and policy instruments

Multiple regulatory and policy instruments exist at international, regional, or national level that either directly target GHG emissions from WWTPs or create incentives that drive reductions (e.g., reporting, methane capture, energy-efficiency). Examples include (i) global voluntary initiatives; (ii) regional regulatory programmes; (iii) national regulatory and reporting programmes; and (iv) market and/or project-based mechanisms.

4.2.1. Global initiatives: The Global Methane Pledge

The **Global Methane Pledge (GMP)**, launched by the European Union and the US Environmental Protection Agency at COP26 in 2021, is a voluntary international commitment to reduce global human-caused methane emissions by at least 30% by 2030, with 2020 levels as the reference point [137]. As of today, 159 countries have joined the initiative, targeting emissions from energy, agriculture, and waste sectors, including wastewater treatment. While the defined target is collective, participating countries commit to take domestic actions across sectors to reduce methane emissions through national action plans and improved emission monitoring using best available inventory methodologies. Furthermore, the Pledge emphasizes cooperation through initiatives like the Global Methane Initiative (GMI) and the Climate and Clean Air Coalition (CCAC). Although non-binding, the Pledge serves as a global framework to accelerate regulation, technology adoption, and investment in methane reduction across sectors.

4.2.2. Regional regulatory programmes

The **Effort Sharing Regulation (ESR)** (in its latest version, Regulation 2023/857 of the European Parliament and of the Council) is part of the EU's implementation of the Paris Agreement and sets binding national GHG reduction targets for each Member State in sectors not covered by the EU Emissions Trading System, including transport, buildings, agriculture, waste, and small industries (all together contributing approximately 60% of EU emissions) [138]. Its goal is to reduce these emissions by 40% from 2005 levels by 2030, with targets tailored to each country. The regulation directly targets methane and nitrous oxide emissions from wastewater treatment, with obligations for Member States to reduce these GHG emissions to meet their national climate targets. The ESR is also expected to contribute to the EU Climate Law's roadmap towards carbon neutrality by 2050 [139].

The revised **EU Urban Wastewater Treatment Directive** (Directive 2024/3019 of the European Parliament and of the Council) defines a number of mandatory targets and actions to reduce GHG emissions from WWTPs, including:

- Energy-neutral operations of all WWTPs treating loads higher than 10,000 PE by 2045
- Identification of measures to increase energy recovery through biogas utilization and heat recovery through energy audits every four years
- Monitoring of GHG emissions (including at least CO_2, N_2O, CH_4) and energy used and produced from all WWTPs treating loads higher than 10,000 PE

Through these measures, the Directive targets a reduction of up 4.86 $MtCO_2e$/y from the wastewater treatment sector in the EU.

4.2.3. National regulatory and reporting programmes

While no country currently has specific regulations that set limits on CO_2 emissions from WWTPs, several countries have introduced instruments used to mitigate CO_2 emissions, such as (i) monitoring and reporting requirements, making emission data public, and (ii) directives (e.g., implementing the EU Urban Wastewater Treatment Directive) or incentives to reduce energy use, increase energy efficiency and enhance the use of renewable energy sources in WWTPs.

An example of the closest thing to a regulatory instrument is the **Climate Plan for a Green Waste Sector and Circular Economy in Denmark**, which in 2020 stated that *"limit values are to be introduced for N_2O emissions from WWTPs treating a load of at least 30,000 PE. These limit values should cover approximately 65% of the total wastewater volume and 75% of the nitrous oxide emissions from the process. Based on the experience gained, discussions with the agreement parties will be held no later than 2025 on whether the threshold should be lowered from 30,000 people (PE) to a smaller scale [140]."* Overall, the goal of the climate plan is to reduce N_2O emissions in Danish WWTPs by 50%. In order to achieve this goal, a proposal has been made to translate the plan into practice through the definition regulatory methods for N_2O mitigation [141]. The proposal introduced (i) recommended limit values, defined as the target EFs to be achieved relative to each WWTP's baseline emissions (i.e. 50% of the baseline N_2O EF); (ii) standardized procedures for monitoring of N_2O emissions through "basic" and "extended" campaigns through certified/approved measurement technologies, for accurate determination of baseline and actual emissions; (iii) the possibility of an N_2O emission tax, calculated based on extra N_2O emissions above the limit. As to (iii), an initial estimate of the shadow price was made to inform the calculation of the tax (i.e., price per CO_2e emitted), showing high variability (from <6 to >1,200 EUR/CO_2e) depending on the type of investment required. The proposal is currently under discussion, requiring the definition of an appropriate shadow price to ensure that the regulation will introduce incentives for N_2O reduction.

The **Greenhouse Gas Reporting Program (GHGRP) of the United States Environmental Protection Agency** defines specific requirements for reporting of methane emissions from selected industrial wastewater treatment facilities (Subpart II). This includes methane generation, recovery and emissions in anaerobic treatment (e.g., bioreactors, digesters and lagoons) and biogas handling systems.

In Australia, the **National Greenhouse and Energy Reporting (NGER) Scheme** defines requirements to measure and report energy consumption and Scope 1 and 2 GHG emissions, resulting in audited and publicly available reports. These requirements apply to WWTPs exceeding 10 TJ/y energy consumption or 25,000 tCO_2e/y emissions.

The **Germany Federal Climate Change Act (Bundes-Klimaschutzgesetz, KSG)** sets binding national GHG reduction pathways and sectoral budgets, which in turn determine sector policy and permitting. Among other stakeholders, water utilities and public authorities must plan investments (energy efficiency, renewables, biogas use) to meet national targets, thereby allowing the prioritization of funding for CO_2 reduction from WWTPs.

4.2.4. Market- and project-based mechanisms

Internationally, other instruments have been established to achieve GHG emissions through voluntary, credit-based schemes.

The most prominent example is the **Clean Development Mechanism (CDM)** under the Kyoto Protocol, which has introduced a framework for upper-middle and high-income countries to invest in emission-reduction projects in low- and middle-income countries and receive Certified Emission Reductions (CERs), i.e. credits that can be used to meet emission reduction targets. Among the available set of projects, CDM specifically targets methane and nitrous oxide reduction in WWTPs, defining methodologies for the calculation of avoided emissions. As an outcome of COP30 (held in November 2025 in Belém, Brazil), CDM will cease operations by the end of 2026, ending its role as Kyoto Protocol's carbon crediting system and transferring remaining funds to the new **Paris Agreement Crediting Mechanism (PACM)**. This decision redirects the focus on Paris Agreement's market mechanisms, with existing CDM projects are expected to transition to PACM to June 2026.

Key takeaways:

- SDG 6.3 highlights the need for safe wastewater treatment, with 44% of household wastewater still untreated globally, and scaling up treatment may increase sectoral CO_2 emissions.
- Global initiatives like the Global Methane Pledge aim to reduce methane emissions from wastewater through voluntary national actions, monitoring, and cooperation frameworks.

- Regional and national regulations (e.g., EU ESR, Urban Wastewater Directive, Denmark's Climate Plan) set binding targets, energy efficiency requirements, and N_2O limits to reduce GHGs from WWTPs.
- Market- and project-based mechanisms (e.g., CDM) provide incentives for voluntary methane and N_2O reduction projects, linking emission reductions to tradable credits.

5. GHG Emissions Mitigation Strategies from Wastewater Treatment Systems

Regulatory instruments and incentives described in the previous section are the drivers to implement GHG mitigation in WWTPs. Actions taken by water utilities and responsible authorities to effectively realize GHG mitigation targets and eventually achieve net-zero emission objectives can be of two types (Figure 5-1):

- **Planning-based approaches,** including holistic system-oriented strategies ranging from major WWTP upgrades to full reconsideration of conventional wastewater management practices
- **Process optimization**, focusing on improving WWTP operations with targeted interventions typically not involving substantial infrastructure changes.

The combination of both approaches, with example strategies and actions described in the following sections, plays a key role in supporting the achievement of national CO_2 emission targets through the contribution of the wastewater treatment sector, and is essential for the implementation of proper planning and implementation of new WWTPs globally in fulfilment of SDG 6.3.

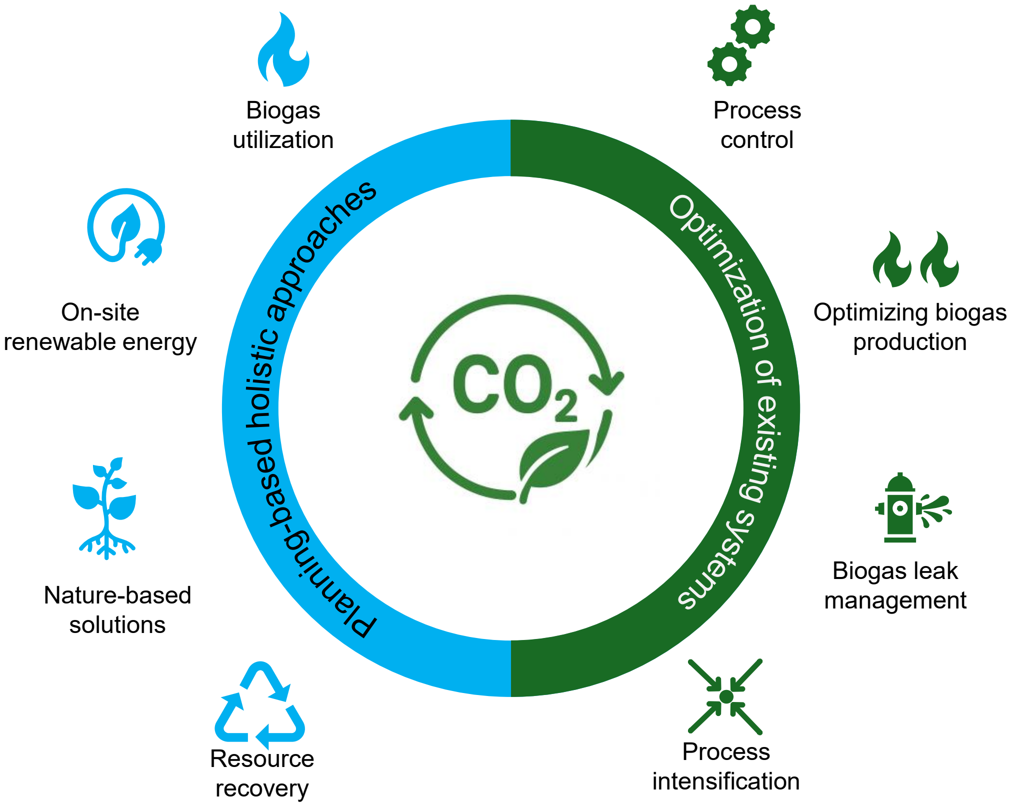

Figure 5-1. Overview of GHG mitigation strategies and solutions for GHG mitigation in WWTPs.

5.1. Planning-based and system-thinking mitigation

By implementing substantial changes in wastewater treatment management through technological transition, planning-based and system-oriented GHG mitigation actions can be effective when a number of factors coexist:

- A favourable political and economic framework, prioritizing climate agendas while supporting innovation and creating economic incentives for relevant stakeholders
- Strategic governance involving all relevant stakeholders, enhancing cooperation between various sectors (e.g., water and energy), setting ambitious, yet realistic goals, and providing clear timelines for their realization
- An open, transparent and cooperative environment for cross-sector stakeholder cooperation, fostering knowledge-, data- and information-sharing while supporting capacity building based on innovation outcomes

5.1.1. Energy neutrality and renewable energy adoption

Renewable energy integration represents one of the most effective paths for reducing indirect greenhouse gas emissions (Scope 2) in wastewater treatment. While direct process emissions, particularly N_2O and CH_4, remain the primary concern, the electricity and heat required for aeration, pumping and sludge treatment are significant contributors to the overall carbon footprint in both centralized and decentralized systems [26, 27]. Transitioning towards energy self-sufficient and net-zero energy facilities requires the strategic adaptation of renewable energy sources, waste-to-energy recovery, and advanced energy management systems.

UNEP has identified the reduction of human-caused methane releases as one of the most cost-effective strategies to reduce global warming [10]. Direct emission of biogas to the atmosphere is ill-advised, since the main component of biogas is CH_4, a more potent GHG than CO_2. Hence, the benefit of biogas utilization is two-fold: (i) combustion converts CH_4 to CO_2, with a significantly lower emissions impact, reducing the overall direct emissions burden (Scope 1); and (ii) using biogas as a fuel provides a renewable source of energy, offsetting the use of fossil fuels and reducing indirect emissions (Scope 2).

Biogas and combined heat and power (CHP) systems. Anaerobic digestion of sludge and organic wastewater fractions generates biogas composed primarily of methane (CH_4), which can be used for combined heat and power (CHP) production. CHP systems allow WWTPs to generate both electricity and thermal energy on site, offsetting fossil fuel consumption and reducing net CO_2 emissions [26, 27, 30, 31]. By capturing and recovering thermal energy, e.g. typically 1-5 times more than when electrical energy is produced, CHP systems have high efficiency (up to 65-80%) [30].

Power and heat recovery from biogas is a globally adopted strategy, not only supporting energy self-sufficiency in WWTPs but also contributing to public energy supply. Brazil's wastewater sector, for instance, has demonstrated that large-scale biogas valorization can provide a major decarbonization lever, especially when paired with grid export and biomethane upgrading [26, 27, 28, 29]. Similarly, several European utilities (e.g., BIOFOS, Copenhagen) have achieved climate-positive operations through optimized biogas-to-energy recovery, enhanced sludge digestion, and use of biogenic CO_2 in local energy systems [25]. In Dehli, India, WWTPs are required to meet at least 60% of their own electricity needs, typically achieved through CHP biogas production [142]. Moreover, excess energy can supply electricity to the grid, fulfilling local power demands. For instance, the McAlpine Creek WWTP in North Carolina, USA generates approximately 7.9 MWh per year; the thermal energy is used to heat the anaerobic digesters on site while electricity is sold to the local energy utility [143].

However, biogas-based energy recovery is not universally feasible for decentralized or small-scale systems, where biogas yield is lower and upgrading infrastructure less economical [9]. Furthermore, CHP systems are available in a large range of sizes (e.g. from tens of kW to hundreds of MW), so they are well-suited to produce decentralized energy and thus avoid transmission losses common in far range utilities.

Key benefit: *Converts on-site waste streams into renewable energy, reducing Scope 1, 2 and 3 emissions while supporting energy self-sufficiency.*

Biogas upgrading and biomethane utilization. Utilizing biogas as a fuel source diverts direct CH_4 emissions that would otherwise be released or flared from anaerobic digestion processes. Biogas can be upgraded to renewable natural gas (RNG) or biomethane, which can then be supplied to the natural gas grid (e.g. as heating and cooking fuel) or as transportation fuel. Thus, biomethane contributes to the energy transition, offsetting fossil fuel consumption. However, to meet these applications, biogas is processed to higher purity standards with a minimum 90% methane composition.

The process of upgrading biogas to biomethane requires the removal of water vapour, CO_2, H_2S and other impurities. Then, gas compressors pressurize the conditioned gas to its final form, either compressed natural gas (CNG) or liquefied natural gas (LNG). The water vapour condensers, membrane purification systems and gas compressors are energy-intensive equipment, producing indirect CO_2 emissions. Furthermore, the activated carbon and polymeric membranes used to remove H_2S and impurities, are derived from coal and fossil-based synthetic polymers. Hence, these materials also contribute indirectly to GHG emissions. Finally, leaks or releases during the storage, upgrading and distribution of biogas and biomethane contribute to direct emissions.

Key benefit: *Converts biogas into renewable energy, reducing Scope 1, 2 and 3 emissions and contributing to decarbonization of the transport sector.*

Case #2: Biomethane utilization

Country: Brazil

Problem: Biogas utilization from animal waste emerged during the oil crisis in 1970s and expanded to recovery from WWTPs in the 1980s. The desire for energy diversification and the increased reliance on renewables continues to spur biogas upgrading and biomethane projects in the 21st century.

Photo credit: SABESP [145]

Solution: A combination of regulatory-driven pilot-projects, policy incentives – e.g., the National Program to Encourage Alternative Sources of Electricity (PROINFA, 2002), the National Biofuels Policy (RenovaBio, 2017) and Zero Methane Programme (Metano Zero, 2022)) – intergovernmental partnerships (e.g., with the German Development Agency, GIZ) and energy net metering have spurred national attention on renewable energy from wastewater, created favorable political and economic frameworks, and facilitated knowledge exchange between institutions to support the rapid technological transition across the energy and sanitation sectors [144]. Since 2018, Sao Paulo's sanitation company (SABESP) has fueled its 40-vehicle fleet with biomethane upgraded from the approximately 3000 m^3 of biogas per day generated at its Franca WWTP (550 L/s) [145]. Looking forward, SANEPAR, a major sanitation company in Paranà state, has initiated a project to recovery biogas for the local production and sale of biomethane (in coordination with United Nations Industrial Development Organization, UNIDO, and the local gas company, Compagas) and has entered a partnership to study the feasibility of a renewable hydrogen plant from the derived biomethane (in coordination with the Brazilian Chamber of Commerce and Industry and German public institutes) [146]. Yet, there are still many obstacles to Brazil to achieving its full biomethane potential, including the need to increase the proportion of wastewater that is safely treated.

Impact: As of 2022, there are 885 biogas facilities in operation, with the sanitation sector representing 10% of the facilities but 74% of the production volume, yet there is significant potential to expand the technology even further [144]. Brazilian Association of Biogas (Abiogás) estimates that 493 million Nm^3/year could be generated from the wastewater sector, which could be further converted to 375 million Nm^3/year of biomethane [147].

Heat and Energy Recovery from Wastewater. Beyond biogas, the wastewater stream itself holds significant untapped thermal potential. Wastewater typically exits households and industries at 10–25 °C, making it a valuable source for heat pumps and district heating systems

[35, 36, 37]. Compared to the chemical energy potential in wastewater influent, the potential thermal energy recoverable from wastewater effluent has been estimated to be 6-8 times greater [148]. The reported electrical equivalent is up to 1.77 kWh/m^3 for heating and 1.18 kWh/m^3 for cooling [149]. Thus, thermal energy offers substantial potential to offset on-site process heating requirements (e.g. sludge drying, anaerobic digestion) as well as providing district heating and cooling to off-site users. While requiring a district heating network for effective use of the recovered heat, this approach has the potential to fully offset GHG emissions from WWTPs [1].

In 2023, Wiener Stadtwerke, Austria's energy and district heating utility, began using heat pumps to harvest 55 MW of heat from wastewater, providing 56,000 households with carbon neutral heating. The utility plans to double the heat pump capacity to supply 56% of households by 2027, reducing the city's CO_2 emissions from heating by 33,000 tons [150]. Similar projects are under development across Europe and Asia, using treated effluent or combined sewer flows as a renewable heat source.

At smaller scales, decentralized heat recovery – via heat exchangers installed in sewer lines or at treatment plant outlets – can provide sustainable heating for nearby communities, reducing energy demand and enhancing local resilience [37]. Such integration can be particularly beneficial where the wastewater system is located close to end users (e.g., residential blocks, industrial parks), minimizing distribution losses.

Key benefit: *Captures wastewater's intrinsic heat energy to offset fossil-based heating demand, lowering overall GHG emissions.*

Photovoltaic (PV) systems. Solar photovoltaic (PV) integration has emerged as a key strategy to further decarbonize WWTPs, particularly by offsetting electricity demand from grid-based fossil energy sources. Given that aeration, pumping, and sludge treatment are among the most energy-intensive processes, the use of on-site renewable generation can substantially reduce Scope 2 emissions and operating costs [152].

Recent large-scale deployment of PV + WWTP systems demonstrates their growing feasibility. Since 2019, hundreds of such projects have been implemented across the Yangtze River Economic Belt in China, where PV installations supply on average 20% of annual electricity demand, leading to an 11% reduction in GHG emissions [152]. The emission reduction potential is strongly dependent on system size, panel coverage, and local solar irradiation. While the initial capital investment of PV systems remains high, it can typically be recovered within 7 years, corresponding to a marginal abatement cost of approximately 26 EUR/t CO2-eq, making PV a cost-effective decarbonization pathway.

Case #3: Wastewater heat pumps

Country: Serbia

Problem: Nearly half of urban households are supplied with district heating, yet fossils fuels provide over 99% of the energy for this heat source. Hence, heating systems have high CO2 emissions and are dependent on energy-imports.

Photo credit: Adobe Stock Photo

Solution: Capturing heat from treated wastewater leaving centralized WWTPs provides a green, local source of energy for nearby district heating systems. A recent Serbian study found that the Šabac WWTP (approximately 50,000 PE) could supply over 20,160 MWh of heat energy, or about 380 kWh per inhabitant [151].

Impact: Applying these figures nation-wide would result in 6.5% decrease in CO_2 emissions from the district heating sector, in addition to diversifying the energy portfolio and reducing import dependence [151].

Similarly, case studies in Europe and the Middle East confirm significant environmental and economic co-benefits. At one domestic WWTP, partial PV integration reduced annual grid electricity consumption by 401,000 kWh, corresponding to a 21% reduction in total carbon footprint and 40% energy cost savings. Full PV coverage was projected to further reduce total emissions by up to 45% [153].

Despite these advantages, PV systems alone cannot fully decarbonize WWTP operations, as electricity demand often exceeds generation capacity during low-irradiance periods. Therefore, hybrid configurations combining PV with biogas-based CHP or battery storage are recommended to ensure energy resilience and optimize renewable integration. Moreover, as global PV deployment increases, life-cycle assessments should account for upstream and end-of-life impacts of PV modules to ensure net-positive environmental performance [152].

Key benefits: *Provides low-maintenance, scalable renewable electricity generation, reducing Scope 2 emissions and operational costs while supporting progress towards energy self-sufficiency and climate-neutral wastewater treatment.*

Further considerations. While individual WWTPs coordinate the on-site use of recovered heat, exporting thermal energy and electricity requires coordination with district heating and energy utilities. Similarly, exporting upgraded biogas (i.e., biomethane or RNG) requires coordination with natural gas utilities and transportation fuel providers. Planning should consider system boundaries, connection points, and construction/maintenance needs before implementation. Understanding the energy market and pricing is another critical element. The revenue that WWTPs receive from the electricity, heat, or biomethane sold supports their maintenance and operations, including wastewater treatment, sludge treatment, and energy generation systems (e.g. biogas production, CHP systems, and heat pumps). Hence, the energy sold should receive a fair price and not be undervalued.

Finally, energy, wastewater and sludge management policies that support and promote sector-coupling to make use of renewable, low-carbon energy sources are a crucial foundation for implementation. Stakeholder involvement, including major corporations and utilities across sectors, from an early stage facilitates the technological transition to using WWTPs as energy producers.

5.1.2. Source separation

Since the 1990s, research institutes and international organizations have continued to explore source separation solutions, yet widespread adoption has been limited due to challenges in implementation. Specifically, urine separation from wastewater is of interest, since it offers the potential to recover valuable nutrients while reducing nitrogen loading to WWTPs.

Urine contributes up to 80% of nitrogen and 50% of phosphorus in domestic wastewater [154], but less than 1% of the total wastewater volume and 6% of COD [155]. Due to the lower influent TN load, the required aeration demand for nitrification falls and the associated energy and indirect emissions decline. Maurer (2006) estimated that treatment of collected urine would require approximately 0.5–2.5 W per person, compared to approximately 4 W per person consumed during conventional nitrification at the WWTP [156]. With less nitrogen, particularly ammonia from hydrolyzed urea, entering the WWTP, the biological treatment process is under less stress, allowing for better COD:N ratios and reducing the risk of N_2O formation. Furthermore, effluent TN is reduced so the potential for N_2O formation in receiving water bodies also declines. Mechanistic modelling simulations have shown that by diverting up to 90% of urine, mechanistic model simulations predict that direct N_2O and CO_2 emissions can be reduced by 98% and 25%, respectively, while indirect CO_2 emissions were reduced by 20% [155].

Since urine is high in both nitrogen and phosphorus, recovered nutrients can be used as a fertilizer, reducing the demand for synthetic fertilizers and emissions associated with their production. In practice, diverted urine is treated by storage or evaporation; several other methods have been explored at the lab-scale including filtration, reverse osmosis, struvite precipitation, acidification, electrochemical treatment, ion exchange and biological treatment based on different treatment goals [156, 157]. However, ammonia volatilization during treatment leads to odours as well as ineffective nitrogen recovery and release to the environment, which must be considered in urine-based fertilizer products' life cycle assessment [157]. Urine also contains a significant portion of organic micropollutants such as hormones and metabolized pharmaceutical byproducts. Hence, urine separation could contribute to more efficient treatment of these contaminants via nanofiltration, electrodialysis or advanced oxidation methods (e.g. ozonation) [156], yet may limit its direct application as a fertilizer depending on local regulations.

Case #4: Urine Division in Durban

Country: South Africa

Problem: Facing a rapidly expanding service area, the eThekwini Municipality introduced urine-diverting (UD) dry toilets to provide sanitation to all citizens because extending centralized wastewater collection and collection systems was impractical due to the hilly landscape and prohibitive costs. The original design routed urine to infiltrate directly into the ground, which posed a risk of polluting local water bodies. Furthermore, users were dissatisfied with the technology, which required household owners to empty the receptacles.

Solution: The eThekwini Water and Sanitation Utility (EWS) reviewed its policies and practices to improve sanitation and the urine diversion programme. In partnership with EAWAG, the Valorization of Urine Nutrients in Africa (VUNA) Project brought together scientists and practitioners to research different technologies for recovering nutrients from diverted urine and using them as fertilizers. The project promoted public acceptance through municipal collection, monetary incentives and health and hygiene education, and operated pilot urine treatment plants for further upscaling [158].

Photo credit: EAWAG [158]

Impact: Over 85,000 UD toilets have been installed, and EWS has continuously updated its UD program, including adaptations to toilet design, providing emptying services, and implementing an intensive public education campaign. While up to 40% expressed satisfaction with the programme [158], several community members have expressed dissatisfaction, viewing the UD toilets as inferior to flush toilets that predominate in other parts of the service area [159]. This criticism highlights the importance of active stakeholder engagement throughout the planning and implementation process to ensure expectations are aligned and that there is public understanding for the final decisions reached. Key take-aways from the VUNA project include the need to demonstrate effective techniques for automated treatment of collected urine so that nutrients can safely be recovered nutrients to produce organic fertilizers; evidence that urine-based fertilizers performed as well as synthetic fertilizers; optimized collection schemes to minimize operational costs and maximize yield; and evaluated practices to improve community satisfaction and use of UD toilets [158].

Source separation implementation requires significant planning for how urine will be separated locally, then collected, treated and applied for beneficial reuse. In existing centralized wastewater systems, urine separation is challenging due to the lack of reasonable collection and transport methods. However, some decentralized systems have practised urine diversion for decades. For example, several communities in Switzerland and Sweden, in both rural and urban areas, have successfully adopted urine diversion (UD) since the 1990s [154, 160]. However, a few systems have discontinued due to poor institutional planning of the collection and reuse of urine; public and farmer disinterest when systems fail to meet promised reuse goals; poorly functioning or outdated UD toilet models; and a lack of economic or legal incentives [161]. Implementing urine separation in specific service areas or neighbourhoods under expansion is one way of introducing urine separation into a centralized wastewater system, but successful adoption must overcome several socio-technical barriers, including ambiguous legal frameworks, lack of profit, low demand, and the high cost of modern UD toilets [160].

Buy-in from the local community is critical for the longevity of urine diversion systems and requires stakeholder engagement, including by residents, local politicians and municipal officials, and farmers (receivers of the collected urine). Norms and regulations around using human excreta, plans for reuse and an explanation of the system as a whole need to be addressed to establish understanding and trust in the source separation efforts. Moreover, public perception that UD toilets produce odours or are inferior to conventional toilets may cause residents to feel marginalized. Finally, UD toilets have different cleaning and maintenance requirements that residents and housing managers must be properly informed about for the system to function.

Key benefits: *Significantly decreases nitrogen loading to the WWTP, thereby reducing aeration energy needs and lowering the potential for* N_2O *emissions.*

5.1.3. Decentralized systems

This section discusses the use of generic decentralized systems as an alternative to centralized treatment, and the potential benefits in terms of GHG reduction. Decentralized nature-based solutions are discussed in greater detail in the following section ("Nature-based and low-technology solutions").

Proximity to population served. Decentralized systems have the potential to reduce indirect emissions from the collection system by minimizing the distance sewage travels to treatment. Decentralized systems are often designed to flow primarily by gravity, thus reducing the pumping requirement in the collection system, reducing energy demand and the need for construction of pumping stations (Scope 2 and 3). Furthermore, localized treatment reduces the need to build large sewer transmission mains, which require significant excavation and construction materials (Scope 3).

Decentralized treatment systems are often located closer to the population, which could benefit from recovered resources, such as heat recovery, biogas for fuel or energy, and beneficial application of treated water and biosolids. Utilization of heat and biogas reduces Scope 2 emissions when used on site as well as direct CH_4 emissions (Scope 1). In the community, heat recovery for district heating, and biogas utilization as a cooking fuel or an energy source, are more economical and efficient when implemented locally. Finally, the location of decentralized systems in rural agricultural areas supports local use of treated sludge (biosolids) and recycled water. Hence, emissions from transporting these resources are lower.

Planning for biogas use. Instead of venting to the atmosphere, biogas collection and utilization is critical for reducing CH_4 emissions from anaerobic technologies (e.g. septic tanks, Imhoff tanks, anaerobic ponds). Burning biogas as a fuel source converts CH_4 to CO_2, which is a less potent GHG. Planning the location of decentralized facilities to maximize biogas production and utilization should consider the population served and its proximity. Biogas recovery will be more economical if installations serve a larger population and where there is more reliable methane production. Where biogas recovery is inappropriate, flaring can be used to convert CH_4 to CO_2.

Covering wastewater facilities. Decentralized WWTPs and other wastewater treatment technologies have a much smaller footprint than centralized systems. The small process area makes them more feasible to cover for emissions capture. However, there needs to be a plan to utilize the captured GHG. While biogas (a mixture of CH_4 and CO_2) provides a fuel source, N_2O has important uses in medicine, dentistry, and the chemical industry (i.e. nitric acid products). It remains to be seen how N_2O emissions from decentralized treatment can be efficiently captured, marketed and purified to industry standards.

Participating in a partnership alliance. Peer-to-peer partnerships between utilities, public operators, or service providers are an excellent opportunity to gain technical knowledge and generally learn from each other's experience in wastewater treatment and GHG mitigation strategies. Decentralized systems, in particular, can benefit from these partnerships as they increase their visibility and advocacy, boost stakeholder engagement, and foster collaboration. For instance, UN Habitat's Global Water Operator Partnership Alliance (GWOPA) facilitates cooperation between utilities to strengthen knowledge-sharing and capacity-building of water and wastewater operators. Historically, GWOPA partnerships have been skewed towards water supply but the organization committed, in its 2023 strategy, to increase engagement in the sanitation sector [162]. By learning "what works" from others' experience, operators of decentralized systems with limited options for local monitoring and testing can still make informed decisions to minimize their carbon footprint.

Case #5: Optimizing biogas recovery from faecal sludge

Country: Burkina Faso

Problem: Ouagadougou relies almost entirely on onsite sanitation systems—septic tanks and pit latrines—generating 500–1,000 m^3/d of faecal sludge [163]. The Kossodo Biogas Plant was built in 2016 to convert this sludge into biogas and electricity, but the achieved biogas production has been considerably lower than what originally planned for. Root causes include, e.g., the low solid content of the received sludge, resulting from the use of water for pit flushing and the collection of greywater inflows.

Solution: Through targeted laboratory scale evaluation of faecal sludge samples, it was found that sludge upconcentration through preliminary settling prior to digestion could lead to a considerable increase of the solids content, resulting in a three-fold increase in biogas yield [163]. As the C/N ratio of the matrix was found to be still below the optimal range, recommendations were made to co-digest the sludge with carbon-rich matrices, such as food waste or agricultural residues,

Impact: This assessment provides actionable recommendations to improve the management of faecal sludge (e.g., reducing water inflow into sanitation pits) and optimize biogas recovery (e.g., through co-digestion with other matrices). The implemented solutions can help achieving the designed biogas production capacity (3,000 m^3/d), thereby reducing greenhouse gas emissions from otherwise sub-optimally managed sludge [163].

5.1.4. Nature-based and low-technology solutions

Proper location. Identifying the proper location for nature-based and low-technology wastewater solutions must consider both population needs and local environmental conditions. Evaluation of the soil conditions and topography is essential to ensure sufficient drainage and prevent waterlogging or flooding. Soil type also impacts soil moisture content tendencies, making it an important factor in oxygen availability for leaching fields, SISs, and CWs. For vegetated technologies, such as CWs and WSPs, sufficient light exposure is essential to sustain algae and plant growth. The local climate and seasonal temperature fluctuations impact plant, algae and microbial growth. In cold climates, the installation of underground or covered technologies can reduce heat loss from wastewater, maintaining higher levels of microbial activity. Decision support tools can assist in evaluating the appropriate technology for a specific site [164].

Regular maintenance. Low-technology and nature-based solutions require regular maintenance and trained operators, even though their operations may not be as complex as conventional WWTPs. Some typical maintenance issues include monitoring the sludge depth in ponds, lagoons, and tanks and periodically removing sludge to maintain hydraulic capacity. Similarly, sludge buildup in infiltration media or CW soils leads to short-circuiting and reduced treatment capacity. Vegetation in CWs requires periodic harvesting to remove plant debris, select specific plant cultures, and maintain optimal pollutant removal. Meanwhile, the growth of unwanted aquatic vegetation in WSPs can impact hydraulic capacity, aeration equipment and light availability for algae. Finally, changes in use of the treatment technology or or population served, including “floating” or migratory populations, may require alterations to normal operations or trigger maintenance activities. Maintaining optimal treatment conditions becomes even more crucial when trying to understand, minimize and control GHG emissions.

Therefore, the maintenance plan is a critical part of nature-based or low-technology systems. Maintenance needs and agency assignment should be determined prior to the installation of a new system or upgrades to an existing system. It is recommended to plan regular maintenance activities at specific intervals and document the agents responsible for fulfilling each task.

Plant selection in construction wetlands. The appropriate plants in CWs and other vegetated technologies must be carefully selected for the local environment conditions for the technology to succeed. Plants are an important source of passive aeration and simultaneously uptake nutrients and pollutants from wastewater. The growth and life cycle of the plant species needs to be considered in the maintenance plan. Initial research also indicates that certain species may reduce N_2O and/or CH_4 emissions from CWs. Where applicable, communities served by vegetated wastewater treatment systems can partner with local universities and research institutions to further understand plant species’ role in minimizing GHG emissions.

Future technologies. Recent developments electrochemically assisted anaerobic digestion show promise into enhancing energy yields while improving pollutant removal. Microbial electrolysis cells (MECs) increase biogas production, reduce H_2S production, and decrease effluent total phosphorus in anaerobic technologies, such as septic tanks. Meanwhile, microbial fuel cells (MFCs) have been shown to generate electricity, providing a local energy source in trickling filters and septic tanks [98, 100]. Further studies are needed to develop these technologies as easy-to-operate units, appropriate for decentralized installations. This is another area where partnerships between communities and local research centres can provide mutual benefits to advance decentralized wastewater treatment and resource recovery.

Case #6: Decision-support tools for the selection of nature-based solutions for wastewater treatment

Country: Global

Problem: Owing to their potential for decentralized domestic wastewater and stormwater treatment, nature-based solutions (NbS) are finding increasing application globally. Clear guidelines for the selection of the most suitable NbS technologies, their design and implementation (also considering local conditions) are generally lacking.

Solution: As part of the Science for Nature and People Partnership (SNAPP), a knowledge-based decision-support tool (SANNAT) has been developed for the selection of most suitable NbS technologies based on local socio-economic and environmental conditions [154], and further support their subsequent design (e.g., required surface area) and implementation. The tool can estimate requirements for, and benefits from, the implementation of NbS, including energy use and carbon sequestration.

Impact: The SANNAT tool provides a detailed, knowledge-based, yet immediate and user-friendly, screening tool for a range of NbS alternatives. Owing to these features, the tool is expected to further the implementation of NbS globally through robust screening and decision-making.

5.1.5. Resource recovery and reuse

A paradigm shift in how we view wastewater is underway. Wastewater is increasingly recognized as a renewable source of heat and energy, carbonaceous materials, nutrients and water [165]. By facilitating the recovery of resources, the wastewater treatment sector can play a role in the circular economy and water-energy-food nexus, and locally support agricultural and industrial sectors to offset their GHG emissions related to the sourcing and transport of water, fertilizers, and polymers.

Water reuse. The main challenge in implementing water reuse comes from public perceptions and regulations. Local regulations around water reuse, or lack of legislation, set the framework for how wastewater providers handle WWTP effluent and the potential for reuse. In particular, direct potable water reuse faces challenges in terms of legislation and public acceptance. Nonetheless, implementation of potable reuse systems continues to spread, including in Namibia, the Philippines, Singapore, South Africa and the United States.

Non-potable reuse, however, is more widely accepted. Non-potable water reuse can fulfil irrigation needs in rural and peri-urban areas as well as the watering of parks, sports fields or golf courses in urban areas. Yet, irrigation demand varies seasonally, so storage facilities (i.e. tanks, reservoirs, cisterns) need to be constructed in addition to distribution piping. Meanwhile, industrial uses of recycled water such as cooling, boiler or process water may vary from day to day but are more stable throughout the year so less storage capacity is needed. Hence, coordination with local agricultural users, industries and community authorities is essential to plan appropriate storage and distribution networks.

Installation of new pipe and pumping infrastructure should be coordinated with other infrastructure projects to minimize excavation and construction climate impacts and costs. Construction emissions from storage facilities must also be factored into the calculations.

Nutrient recovery. The recovery of nutrients, such as phosphorus and nitrogen, provides an alternative to synthetic chemical fertilizers. Local recovery from wastewater treatment facilities offsets emissions from the mining, transport, and production of synthetic fertilizers. For instance, phosphorus rock, though limited, is the main abiotic resource for synthetic fertilizers. Mined phosphorus is exported across the globe. Meanwhile, the energy-intensive Haber-Bosch process is currently the main pathway to synthesize ammonia-based fertilizers. Hence, at a national level, the ability to offset emissions from the agricultural sector should be weighed against the additional emissions incurred by the wastewater sector in implementing nutrient recovery measures. The impact of released nutrients in wastewater effluent must also be considered. Nutrient-rich discharges contribute to the eutrophication of water bodies. Moreover, microbial processes, similar to those found in activated sludge treatment and anaerobic digestion, continue in nature, leading to the potential formation and release of direct GHG emissions from water bodies or soils. Hence, effluent nutrients are considered as part of the off-site emissions of wastewater treatment facilities.

Various technologies are available to recover **phosphorus** from wastewater or sewage sludge. Recovery technologies provide a local source of phosphorus, reducing the mineral depletion and the emissions associated with mining, transporting, and producing P fertilizers from phosphorus rock. These recovery technologies, therefore, have the potential for offsetting GHG emissions from the agricultural sector or other sectors dependent on phosphorus. Phosphorus recovery technologies include:

- **Recovery from solid streams:** Wet-chemical and thermochemical recovery methods are applied to recover phosphorus from dewatered or dried sewage sludge or incinerated sludge ash [99, 122]. Thermochemical recovery, in particular, is characterized by reduced chemical requirements and waste generation, resulting in potentially lower Scope 3 emissions, and can offset energy requirements through heat recovery.
- **Recovery from liquid streams:** Chemical precipitation, such as struvite or calcium phosphate precipitation, is a well-established technique, especially from concentrated process streams from sludge dewatering. Used globally, precipitation is associated with chemical and energy use, thereby resulting in not negligible GHG emissions. Adsorption of phosphorus from liquid streams may reduce both energy consumption and chemical additives, and certain adsorbent materials (e.g. crab carapace micropowder CCM) have shown promising results, though they are applicable only to low turbidity streams (e.g. secondary effluent or filtrate) [123]. Use of bio-adsorptive materials, or further reuse of adsorbents as soil amendments, are possible solutions for offsetting emissions associated with the production, regeneration and disposal of sorbent materials.

In addition to struvite precipitation, **recovery of nitrogen** from highly concentrated stream generated from sludge handling (e.g., reject water) can be achieved through various technologies including ammonia stripping combined with acid absorption, gas-phase ammonia transfer, and membrane or sorption-based separation [165]. In particular, ammonia stripping and absorption (e.g., using a packed-tower gas–liquid transfer to form ammonium sulfate) have reached full-scale application and are capable of removing >90% of NH_4-N from concentrated streams. Though energy and chemical demands pose a challenge, ammonia stripping can provide a double benefit in terms of GHG emission reduction, i.e. (i) reduction of N_2O emissions as compared to biological main- and sidestream treatment of reject water, and (ii) generation of marketable fertilizer products, displacing synthetic fertilizers.

Finally, algae and cyanobacteria have shown potential for removing phosphorus and nitrogen from wastewater. While these types of technology require energy to meet light intensity and temperature needs as well as pumping, filtration, and drying of harvested algae, algae monocultures have even shown promise in terms of replacing secondary treatment at small-scale WWTPs [124], thereby eliminating energy-intensive aerobic treatment and its associated emissions.

Carbon and Added Value Product Recovery. Wastewater bacteria can produce valuable carbonaceous materials, such as biopolymers, bioplastics, biochars and green graphene, contributing to the circular economy. In the Netherlands, a demonstration plant has been operational since 2022, using wastewater bacteria to produce polyhydroxylalkanoate (PHA), a type of bioplastic that can be used in agricultural and horticultural applications as well as in self-healing concrete [166]. Other studied uses of biopolymers include adhesives, corrosion inhibitors, coatings, fire retardants, flocculants, adsorbents and even cosmetics [167, 168]. Biopolymers, in particular, present an opportunity to replace conventional polymers synthesized in the petrochemical industry, reducing reliance on fossil fuels [169]. The substitution of synthetic polymers with sustainable alternatives could offset the carbon footprint of materials in several industrial sectors, including the water and wastewater sector which relies on flocculants and adsorbents in treatment technologies.

Moreover, wastewater-based biomaterials are more sustainable alternatives to both synthetic materials and plant-based biomaterials. In general, biomaterials present an opportunity to reduce reliance on fossil-based polymers and carbons synthesized in the petrochemical industry. Yet mass production of plant-based biomaterials poses other environmental concerns, such as competition for land with food production or threats to natural biodiversity due to the harvesting of key species [169]. Conversely, wastewater-based biomaterials derive valuable resources from a product traditionally viewed as waste. A substantial paradigm shift and behavior change will be needed to move from fossil-based materials to wastewater-based biomaterials. While requiring a change in public perception, valorization depends on the existence of a market for the extracted biomaterials, and a regulatory framework can help to shape the market for biomaterials. Material standards for new materials should be established to ensure the quality of bio-alternatives. New policies restricting the use of plastics in agriculture, horticulture and other environmental sectors, aimed at reducing microplastic pollution, could incentivize further development of biopolymers.

5.1.6. The role of life cycle assessment

Life cycle assessment entails a comprehensive analysis of both the upstream and downstream impacts of a WWTP process or product throughout its entire life cycle. For GHG emissions accounting, energy consumption and other emissions-related environmental impacts are specifically considered. For example, introducing an activated carbon treatment process at the WWTP needs to consider the energy and emissions from the extraction and production of the activated carbon as well as the impacts of regenerating or disposing the spent media after its useful life. As emissions accounting moves towards including Scope 3 emissions (currently voluntary reporting), the role of LCA will become increasingly important.

There are several standards, methods and tools available that decision-makers and utility managers can use to guide and assist with life cycle analysis related to WWTP emissions accounting. The International Standards Organization (ISO) defines the principles and framework for LCA in ISO 14040:2006 while specifying requirements and provides guidelines for LCA in ISO14044:2006.

Conducting an impact assessment (LCIA) is an essential component of the LCA, with the purpose of translating emissions and resource extractions into comparable environmental impact scores. Two methods are widely accepted for LCIA – the GLAM (Guidance for LCA indicators and Methods, last updated in 2024) and the ReCiPe method (last updated 2016). Both methods use a system of characterization and weighting factors to arrive at comparable impact scores across Areas of Protection. These methods are the product of scientific consensus and collaboration between institutions and stakeholders globally. Several tools are available to assist in the LCA and LCIA, including SimaPro Craft software, Environmental Assessment System for Environmental TECHnologies (EASETECH), and the Ecoinvent database.

LCA-based methods have been used to determine the total CO_2 footprint, and relative contributions from various sources (direct and indirect emissions), for a WWTP [92, 93]. In addition, LCA has been applied to assess the benefits (also in terms of reduced CO_2 emissions) of implementing new wastewater management approaches, e.g. mainstream nutrient recovery [170] and handling of digestate [171].

5.2. Process optimization

While planning-based, holistic approaches are essential for ensuring sustainable wastewater management practices and long-term mitigation of GHG emissions, targeted interventions for optimizing existing process operations can achieve tangible, substantial outcome – particularly in the reduction of direct CH_4 and N_2O emissions. Optimization actions typically aim at identifying process bottlenecks (e.g., leaks) and/or improve the operation of specific processes through targeted interventions not requiring major investments.

5.2.1. Process optimization and control for optimizing CH_4 recovery

Strategies for management of sludge and utilization of biogas determine biogas conditioning needs and appropriate process and storage equipment. Optimizing these processes is unique to the individual WWTP and its biogas programme. Yet, regular performance audits can indicate the effectiveness of biogas utilization and identify potential areas for improvement. As an example, CH_4 destruction in a low-efficiency flare can be less than 60%, so upgrading equipment has great potential for emissions reduction [2]. Selecting the appropriate technology to beneficially use produced biogas for in situ electricity cogeneration contributes to significant reductions in facility-wide GHG emissions [172]. A number of solutions have demonstrated potential, through full-scale WWTP applications, for increasing biogas production.

Enhancing biogas production. While dewatered primary and secondary sludge are efficient substrates for biogas production from anaerobic digestion, various strategies have emerged to enhance biogas production from anaerobic digesters.

A commonly adopted strategy is the co-digestion of sewage sludge with other solid matrices or slurries rich in organic content, including food waste, oil-grease and fish sludge [166]. Full-scale examples have shown significant increases in biogas production as a result of co-digestion, allowing energy-positive WWTP operations through power and heat recovery from biogas to be achieved. Conducting biomethane potential tests provides crucial information to assess process feasibility and to optimize performance, particularly when implementing co-digestion strategies with diverse sludge types or substrates.

Meanwhile, pre-treatment technologies are increasingly applied to increase the biogas production from sewage sludge. Prominent examples of pre-treatment technologies include:

- **Thermal hydrolysis process (THP)**, whereby sludge is subject to high temperature and pressure to generate soluble products, with faster conversion to biogas. Through this process, a sanitized and more easily dewaterable sludge is produced, thereby leading to more efficient disposal and safer reuse. Conversely, this process also results in ammonia release, requiring efficient treatment of more concentrated reject water or potential coupling with nitrogen recovery technologies.
- **Vacuum processes,** whereby sludge is exposed to low pressure conditions, disrupting cell walls and flocs through water vaporization and gas bubbles formation inside the sludge matrix. Collapsing bubbles upon pressure release enhance cell lysis, with release of organics and nutrients in solution. Similarly to THP, this process helps achieve higher biogas yield and improved dewaterability, with reduced energy use. As a separate pre-treatment, or integrated into digestion units [167], vacuum processes can also support process intensification by reducing digester volumes [168].

Case #7: Combining system-based and optimization approaches for net zero goals in water utilities

Country: United Kingdom

Problem: Thames Water, UK's largest water utility, has identified direct GHG emissions of methane and nitrous oxide and indirect emissions from energy use as the main contributors to its CO_2 footprint. The utility aims to achieve national net-zero commitments by 2050 by reducing GHG emissions and converting waste streams into low-carbon energy assets.

Solution: Thames Water has defined a strategy to achieve this goal through four best management practices, combining long-term investments and incentives for new technologies with continuous optimization and maintenance: (i) prioritizing methane capture through centralized large-scale anaerobic digestion; (ii) enabling biogas conversion to energy through CHP for on-site power and heat use in WWTPs, and biogas upgrading to biomethane (iii) optimization of biogas production through pre-treatment (e.g., thermal hydrolysis); and (iv) integrating carbon targets into business planning as clear corporate targets, prioritizing implementation of impactful projects while securing funds for their implementation.

Impact: The implementation of prioritized actions is expected to help Thames Water achieve (i) a 17% increase in biogas yields, combined with improved sludge dewaterability and more efficient disposal, through thermal hydrolysis; (ii) annual self-generation of approximately 317 GWh of electricity; (iii) operational resilience through reduced exposure to wholesale electricity and gas price volatility, thereby lowering operating costs of WWTPs.

5.2.2. CH_4 leak management

Methane gas detection system. In addition to being a potent GHG, the combustibility of CH_4 makes it a safety concern for plant staff. Implementing a methane detection system, that can track gas levels and alert staff in case of danger, is an essential element of on-site safety, particularly for WWTPs that collect and use biogas. As a minimum, fixed CH_4 gas detectors should be installed in occupied buildings and enclosed work areas associated with influent wastewater/headworks, anaerobic digestion, sludge storage and handling, and biogas storage and processing. While the main purpose of the detection system is to maintain a safe working environment, sudden changes in CH_4 gas levels may be indicative of a leak and justify further investigation.

Regular leak monitoring. Regular monitoring ensures that leaks are quickly detected and repaired. Ideally, continuous monitoring systems should be implemented to improve the speed of detection. The implementation of advanced monitoring technologies, such as remote sensors or gas imaging cameras, can further improve accuracy and identification of unknown leaks. Sensors and cameras should be strategically located to maximize site coverage and detection efficiency, for example focusing on critical process areas such as anaerobic digestion and sludge handling. Sensors must be regularly calibrated to maintain accuracy.

Periodic audits of the monitoring system are recommended to identify potential issues and improve leak detection. Similarly, regular leak monitoring campaigns should be conducted, where continuous monitoring is not feasible. To this end, remote or mobile gas detectors are typically used to identify CH_4 leaks from tanks, vents, piping, and other hard-to-reach process areas (see “Monitoring of GHG emissions”). Fixes and adjustments resulting from routine monitoring have been shown to halve fugitive methane emissions [105].

Operations personnel should receive routine training in the monitoring processes to ensure proficiency in sensor use, calibration and repair or replacement. Training should include the documentation process to ensure consistency of data collection, including monitoring data, detailed records of leak detections and responses, and system auditing. Comprehensive documentation facilitates GHG emissions accounting and regulatory reporting.

Cover and capture systems. Influent to the WWTP and sludge storage areas are other common sources of CH_4 emissions, that often are not covered. Enclosing these areas would improve emissions monitoring and the potential to capture and utilize released CH_4 that would otherwise be vented to the atmosphere.

Case #8: Methane capture from covered sludge storage and vacuum capture systems

Country: Denmark

Problem: The Ejby Mølle WWTP (Odense, Denmark) identified methane emissions from open-air sludge storage as a considerable contributor to the CO_2 footprint of the WWTP.

Photo credit: Miljøstyrelsen [12]

Solution: The WWTP has implemented a new closed sludge storage tank with gas collection, directly connected to the biogas system. To further enhance gas utilization, a vacuum system was installed to recover dissolved methane before it is released, turning a previously uncontrolled emission into a valuable resource.

Impact: The closed tank led to a measured 45% reduction in methane emissions, and when accounting for emissions from the gas engine, the total reduction reached 56% across the entire plant [12]. An LCA comparing steel versus concrete tank construction showed a 7 kg CO2-eq savings per m^3 of stored sludge with steel. Additional benefits included improved dewatering of digested sludge, with dry solids content increasing by 1–2%, resulting in annual cost savings of up to 80,000 EUR.

5.2.3. Process optimization and control for N_2O reduction

N_2O emissions represent the majority of GHG emissions for most WWTPs. However, it is estimated that up to 56% of N_2O emissions can be mitigated in the short-term by applying process optimization actions [2]. Such measures are most effective for centralized WWTPs and activated sludge-based decentralized systems, as N_2O emissions typically represent a small share of total GHG emissions in nature-based or less mechanical decentralized wastewater treatments [173].

Effective mitigation of N_2O emissions in WWTPs requires process control strategies that address the operational and environmental conditions responsible for N_2O formation. As demonstrated in the previous section, dissolved oxygen (DO), nitrite accumulation, transient load changes, substrate composition, and pH–temperature dynamics are major determinants of emission behaviour [33, 43, 46, 174]. Process optimization therefore focuses on regulating these key factors through targeted operational adjustments and advanced control systems.

Improving aeration and dissolved oxygen control. Due to the strong influence of dissolved oxygen (DO) concentrations, implementing improved aeration control strategies is one of the most effective measures to reduce N_2O emissions. DO plays a central role in both nitrification and denitrification processes, and its optimal management can minimize N_2O formation by avoiding nitrite (NO_2^-) accumulation and incomplete denitrification. Advanced aeration control strategies such as intermittent aeration, step aeration or nutrient-based aeration control can substantially reduce both energy use and N_2O generation, mitigating both direct and indirect emissions [175].

Intermittent aeration has been shown to simultaneously reduce N_2O emissions and energy requirements, thereby mitigating both direct and indirect emissions [175] [176]. Intermittent aeration helps prevent the buildup of nitrite, reducing N_2O generation by up to 90% [173] [175]. Dynamic aeration strategies that maintain low, but stable DO concentrations are particularly effective in balancing nitrification efficiency and N_2O mitigation. For example, maintaining a DO setpoint around 0.5 mg/L allows for simultaneous nitrification and denitrification within the same reactor zone, minimizing nitrite accumulation and promoting complete nitrogen removal. Implementing a setpoint of 0.5 mg/L in a full-scale Sequencing Batch Reactor (SBR) system reduced the N_2O emissions factor by 35% [176]. However, careful calibration of DO levels is crucial. Too low DO inhibits ammonia oxidation and promotes N_2O generation via the nitrifier denitrification pathway, while too high DO inhibits denitrification and can increase N_2O stripping from the liquid phase.

Upgrading to more advanced control systems typically requires investments in operator training, online sensors, and automation, but often results in improved aeration efficiency and reduced indirect CO_2 emissions. It is generally more challenging to control N_2O emissions in systems with high variable influent characteristics [109]. Therefore, the type of aeration and equipment selection should be critically evaluated to promote even air distribution and to prevent large spatial variations in gas transfer.

Adding artificial aeration to decentralized treatment technologies that traditionally rely on passive aeration e.g. Soil infiltration systems (SISs) and Constructed Wetlands (CWs), may also be appropriate for installations with significant GHG emission rates. However, the benefits of aeration control in these systems should be weighed against the addition of indirect emissions from operating blowers.

Key benefit: *Reduces* N_2O *while simultaneously achieving required nutrient removal through better management of aeration requirements, with additional reduction of energy-related* CO_2 *emissions.*

Managing C:N ratio and feeding schemes. The carbon-to-nitrogen (C:N) ratio strongly influences denitrification completeness. Operating within optimal ranges (C:N = 4-6) promotes full reduction of NO_3^- and NO_2^- to N_2. In centralized WWTPs, this is achieved by maintaining sufficient COD in anoxic zones, often through external carbon dosing or from a side stream of COD-rich influent. The carbon source itself influences emissions, e.g., dosing of acetate was found to lead to lower N_2O emissions than methanol, but higher than ethanol, glucose or soluble starch [48, 175].

The C:N ratio changes across the reactor unit as organics and nutrients are consumed in biological treatment. Carbon is rapidly consumed by biomass under aerobic conditions, meaning the wastewater may be carbon limited when it reaches anoxic zones. Introducing step or intermittent feeding schemes can enhance COD utilization, thereby preventing the accumulation of NO_2^- and N_2O. Lab-scale studies have shown up to a 66% reduction in N_2O emissions with optimised feed control [173].

In decentralized technologies, organic content can be artificially enhanced by adding biochar as a source of slowly degradable carbon. This approach has been tested in septic tanks for increased biogas production as well as in SISs and CWs to reduce N_2O emissions [98, 100]. However, biochar addition adds to indirect emissions under Scope 3, as the production of biochar is energy intensive, requiring pyrolysis (high heat under oxygen-limited conditions), and chemical treatment, of organic material. Biochar can also be produced from wastewater sludge, presenting an opportunity for local sourcing.

Key benefit: *Improves denitrification efficiency and carbon utilization while lowering* N_2O *formation.*

Flow equalization. Implementation of flow equalization can mitigate daily TN and ammonia loading peaks. Since N_2O emissions are associated with high ammonia (NH_3) and nitrite (NO_2^-) concentrations, removing peaks in nitrogen loading to WWTPs should reduce N_2O emissions and provide more stable biological conversion rates. As a result, N_2O emissions would exhibit less variability, facilitating their control.

Key benefit: *Reduces peak* N_2O *emissions and facilitates stable process operation.*

Control of microbial populations. The bacterial populations present in the biomass and their relative quantities will affect the conversion rates and pathways, thereby influencing the amount of N_2O generated and consumed during wastewater treatment. For instance, increasing the amount of ammonia oxidizing bacteria (AOB) decreases the overall ammonia oxidation rate and thereby N_2O generation. Mathematical modelling indicates that changing the returned activated sludge (RAS) return scheme to return to two locations promotes AOB growth and could reduce N_2O emissions by up to 50% [177]. Another reported strategy relies on a longer solids retention time (SRT) to promote the growth of nitrite oxidizing bacteria (NOB), which consumes NO_2^-, an important precursor to N_2O generation [178].

Researchers have also identified several nitrifying organisms (e.g. ammonia oxidizing archaea (AOA), acid-tolerant AOB *Candidatus nitrosacidococcus & C. nitrosoglobous*, and *Nitrospira*), whose unique nitrogen metabolism characteristics preclude or significantly reduce the generation of N_2O compared to traditional nitrifying bacteria [107]. Additionally, the selection of organisms that uptake ammonia-nitrogen directly, thereby avoiding nitrification and N_2O generation, deserves further consideration (e.g., algae, archaea, fungi, phototrophic bacteria).

Key benefit: *Long-term N_2O mitigation through strategic microbial community control.*

pH control. The system's pH affects the optimal growth of biomass. The NH_2OH oxidation pathway can be controlled by restricting AOB growth by maintaining a pH of 6-7 [173]. On the other hand, denitrification and N_2O reduction can be inhibited by low pH. To ensure complete nitrification and N_2O consumption through denitrification, it is recommended to control pH at around 7 and temperature at 20°C [175].

Key benefit: *Stabilizes biological processes and limits N_2O formation from both nitrification and denitrification.*

Off-gas treatment. Where wastewater treatment processes can be covered to collect off-gas, there is the possibility to treat the off-gas for N_2O before release. Off-gas treatment options include co-combustion with methane (or biogas) on-site or conversion to N_2 gas via biofilters or bioscrubbers [173]. This strategy is more viable for small mechanical, activated sludge plants where the reactor footprint is minimal. However, these technologies have not been tested for full-scale application yet.

Key benefit: *Enables end-of-pipe control of residual N_2O emissions.*

Summary. Process optimization for N_2O reduction requires a holistic operational control approach integrating aeration, pH, temperature, and load management. By maintaining stable conditions, optimizing DO and C:N ratios, and preventing process transients, WWTPs can significantly reduce both the frequency and intensity of N_2O emission events. When combined with gas collection and off-gas treatment systems, these operational measures form the foundation for achieving low-emission wastewater treatment.

5.2.4. Process Intensification

The dilemma of process intensification arises when operational changes that were meant to make processes more efficient result in unintended increases in GHG emissions. Therefore, it is important to evaluate process intensifications holistically and determine how the process contributes to the overall GHG emissions inventory.

Intensification of biological nitrogen removal (BNR) aims to meet treatment goals despite higher nutrient loading or higher flows. Yet, these conditions (e.g. high NH_4 and TN concentrations) are correlated with N_2O peaks. While the N_2O EF from the main biological treatment process average 1.01% kg N_2O-N/kg TN, N_2O EFs from sidestream treatment, where the nitrogen removal rate is more intense, are higher, with mean of 2.82% [26]. Advanced BNR processes (e.g., nitritation-denitritation or partial nitritation-annamox) have lower oxygen and external carbon source demand with a faster conversion rate compared to conventional BNR (i.e., nitrification-denitrification process) and thereby require less energy and smaller tank volume. Although Scope 2 and 3 emissions may be reduced by adopting advanced BNR, reports of increased N_2O emissions from these processes may render the overall C-footprint higher than a conventional process [175].

Strategies to intensify biogas production may implement sludge pre-treatment processes to improve sludge digestibility. The result is more biogas and reduced digested sludge volume, but also increased nutrient release during anaerobic digestion, necessitating additional chemicals (e.g. methanol as an external carbon source) to support nutrient removal in the main WWTP. For instance, implementation of a thermal hydrolysis process (THP) ahead of the anaerobic digestors at Blue Plains Advanced Water Treatment Plant (Washington, DC, USA) resulted in a 54% increase in methanol consumption to support denitrification in the activated sludge process, contributing an estimated 60 to 85% of the plant’s Scope 1 emissions as a source of anthropogenic CO_2 [179]. Unlike biogenic CO_2 released from the treatment of raw sewage, methanol is a fossil-fuel-derived chemical and its conversion to CO_2 during wastewater treatment must be quantified as an anthropogenic release.

6. Concluding remarks and recommendations

Direct and indirect emissions of greenhouse gases from wastewater treatment plants are substantial contributors to the global CO_2 footprint of the water sector. While actions aimed at quantifying the magnitude of GHG emissions have increased in the past decade, considerable uncertainties in the estimates persist. Recent findings and ongoing investigations increasingly show that evaluations made so far may have underestimated the magnitude of GHG emissions, making monitoring at WWTP-, national- and regional-scale even more urgent.

Nevertheless, available information has aided a good understanding of the main contributors of GHG emissions from WWTPs, helping to direct and prioritize mitigation actions. This knowledge is especially important in the proper planning of new wastewater treatment facilities to be built in an effort to improve sanitation (SDG 6.3), achieving low GHG emissions from the early stages of a facility's life cycle.

For both already operating and new wastewater treatment facilities, we have summarized a set of actionable recommendations for GHG mitigation (Table 6-1) which aim to support decision-makers and practitioners (operators, engineers, designers) in identifying the most suitable way to achieve net-zero wastewater treatment while preserving sanitation goals.

Table 6-1. Summary catalogue of planning-based and optimization-based mitigation actions.

Type	Action	Targeted / avoided GHG emissions	Benefits	Costs CapEx / OpEx	Suitable for Centralized (C), Decentralized (D) systems
Planning ***Energy Neutrality***	Energy recovery from biogas using CHP	Scope 1 (CH_4) Scope 2 (energy) Scope 3 (biosolids disposal)	Conversion of waste streams into renewable energy, reducing GHG emissions while supporting energy self-sufficiency	++ / ++	C
Planning ***Energy Neutrality***	Biogas upgrading to biomethane	Scope 1 (CH_4) Scope 2 (energy) Scope 3 (biosolids disposal)	Conversion of biogas into renewable energy, generating biomethane and thus contributing to decarbonization of the transport sector	++ / ++	C, D
Planning ***Energy Neutrality***	Heat recovery from wastewater	Scope 2 (energy)	Conversion of wastewater's intrinsic heat energy to heat for residential use, offset fossil-based heating demand and potentially fully offsetting GHG emissions from WWTPs	++ / ++	C, D
Planning ***Energy Neutrality***	On-site photovoltaic systems	Scope 2 (energy)	Provides low-maintenance, scalable renewable electricity generation to support path towards energy self-sufficiency	++ / ++	C, D
Planning ***Source Separation***	Urine diversion	Scope 1 (N_2O) Scope 2 (energy) Scope 3 (fertilizer use)	Decrease of nitrogen loads to WWTPs, with reduction in N_2O emissions and aeration power	+++ / ++	D

Planning ***Decentralized treatment***	Implementation of decentralized treatment for small, remote communities	Scope 1 (N_2O, CH_4) Scope 3 (infrastructure)	Significant reduction in infrastructure for wastewater collection	+++ / ++	D
Planning ***Decentralized treatment***	Implementation of decentralized nature-based solutions	Scope 1 (N_2O, CH_4) Scope 3 (infrastructure)	Reduced footprint, with possibility for more efficient capture of emissions	+++ / ++	D
Planning ***Resource recovery***	Nitrogen recovery from highly concentrated streams (e.g., reject water from sludge handling)	Scope 1 (N_2O) Scope 2 (energy) Scope 3 (fertilizer use)	Reduced N_2O emissions and aeration energy use from reject water stream load reduction, and recovery of ammonia as fertilizer material	+++ / ++	C
Planning ***Resource recovery***	Recovery of phosphorus from solid (e.g., struvite) and liquid streams for fertilizer production	Scope 3 (fertilizer use)	Reduced reliance on mineral phosphorus as fertilizer	+++ / ++	C
Planning ***Resource recovery***	Recovery of added value organics (e.g., biopolymers)	Scope 3 (material use)	Reduced reliance on fossil-derived materials (e.g., chemicals, plastics)	+++ / ++	C
Planning / Optimization ***Increasing biogas production***	Pre-treatment technologies (e.g., thermal hydrolysis) and co-digestion strategies	Scope 2 (energy) Scope 3 (biosolids disposal)	More effective conversion of organics in sludge to biogas, with increased energy self-sufficiency and reduced volumes of high-quality biosolids requiring disposal	+++ / ++	C
Optimization ***Leak reduction***	Monitoring devices to detect potential leaks and regular maintenance of equipment (e.g., piping) to reduce leaks	Scope 1 (CH_4)	Reduction of undesired fugitive emissions and mitigation of risks for operation staff	++ / ++	C, D
Optimization ***Cover and capture***	Covering of bioreactors and sludge storage areas and collection / treatment of off-gas	Scope 1 (N_2O, CH_4)	Utilization of residual CH_4 and abatement of N_2O emissions from off-gases. Capture of CO_2 from off-gases for downstream applications	++ / ++	C, D

Optimization ***Process control for targeted N_2O reduction***	Automated dynamic aeration strategies (intermittent aeration, ammonia-based aeration control)	Scope 1 (N_2O) Scope 2 (energy) Scope 3 (effluent N_2O)	Reduction of N_2O emissions while achieving required nutrient removal, with additional reduction of energy-related CO_2 emissions by avoiding over-aeration	+ / +	C, D
Optimization ***Process control for targeted N_2O reduction***	Optimal C-to-N ratio through step-feed or external carbon dosing to	Scope 1 (N_2O) Scope 3 (effluent N_2O)	Improved denitrification, acting as a sink for N_2O	+ / +	C
Optimization ***Process control for target N_2O reduction***	Flow equalization to buffer high influent NH_4-N loads	Scope 1 (N_2O) Scope 3 (effluent N_2O)	Stable operation and reduced N_2O emissions through buffering of peak loads	++ / +	C

References

1. Polesel, F. and T. Dalkvist, *Analysis of the potential contribution to energy and climate neutrality from Danish technology within the global wastewater sector*. 2022.
2. Song, C., et al., *Defining and achieving net-zero emissions in the wastewater sector.* Nature Water, 2024. **2**(10): p. 927-935.
3. Lambiasi, L., et al., *Greenhouse gas emissions from sanitation and wastewater management systems: a review.* Journal of Water and Climate Change, 2024. **15**(4): p. 1797-1819.
4. Dickin, S., et al., *Sustainable sanitation and gaps in global climate policy and financing.* npj Clean Water, 2020. **3**(1): p. 24.
5. IPCC, *Climate Change 2023: Synthesis Report for the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. 2023: Geneva, Switzerland.
6. de Vries, W., *Impacts of nitrogen emissions on ecosystems and human health: A mini review.* Current Opinion in Environmental Science & Health, 2021. **21**: p. 100249.
7. EPA, *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2021*. 2023.
8. Tian, H.A.-O., et al., *A comprehensive quantification of global nitrous oxide sources and sinks*. Nature, 2020. **586**(7828): p. 248-256.
9. Zhu, G., et al., *Nitrous oxide sources, mechanisms and mitigation*. Nature Reviews Earth & Environment, 2025. **6**(9): p. 574-592.
10. UNEP and CCAC, *Global Methane Assessment: Benefits and Costs of Mitigating Methane Emissions*. 2021: Nairobi.
11. EEA. *Greenhouse gas emission intensity of electricity generation in Europe*. 2025 [cited 2025 20 October]; Available from: https://www.eea.europa.eu/en/analysis/indicators/greenhouse-gas-emission-intensity-of-1?activeAccordion=.
12. Rebsdorf, M.P., Line Dybdahl; Fredenslund, Anders; Scheutz, Charlotte; Thomsen, Marianne; Tornberg, Dines; Eriksson, Michael; Mielczarek, Artur; Vangsgaard, Anna Katrine; Madsen, Jeanette Agertved; Askjær, Niels Malmmose; Nielsen, Per Henrik; Hansen, Rikke; Knudsen, Alma Skjold; Almind-Jørgensen, Nina *ARES Aktiv Reduktion af Emissioner fra Spildevandsanlæg - MUDP-Projekt*. 2024. p. 124.
13. Law, Y., et al., *Fossil organic carbon in wastewater and its fate in treatment plants*. Water Research, 2013. **47**(14): p. 5270-5281.
14. Morales-Rico, P., J. Ramos-Díaz, and F. Thalasso, *Greenhouse Gas Mass-Balance in Conventional Activated Sludge Wastewater Treatment: A Case Study in Mexico for Developing Countries.* ACS Omega, 2025. **10**(6): p. 5574-5581.
15. Paredes, M.G., et al., *Methane emissions from stabilization ponds for municipal wastewater treatment in Mexico.* Journal of Integrative Environmental Sciences, 2015. **12**(sup1): p. 139-153.
16. Silva, J.P., et al., *Influence of photoperiod on carbon dioxide and methane emissions from two pilot-scale stabilization ponds.* Water Science and Technology, 2012. **66**(9): p. 1930-1940.

17. Detweiler, A.M., et al., *Characterization of methane flux from photosynthetic oxidation ponds in a wastewater treatment plant.* Water Science and Technology, 2014. **70**(6): p. 980-989.
18. Hernandez-Paniagua, I.Y., et al., *Greenhouse gas emissions from stabilization ponds in subtropical climate.* Environmental Technology, 2014. **35**(6): p. 727-734.
19. Gruber, W., et al., *Estimation of countrywide N2O emissions from wastewater treatment in Switzerland using long-term monitoring data.* Water Research X, 2021. **13**: p. 100122.
20. Gruber, W., et al., *Linking seasonal N2O emissions and nitrification failures to microbial dynamics in a SBR wastewater treatment plant.* Water Research X, 2021. **11**: p. 100098.
21. Gålfalk, M., et al., *Ground-based remote sensing of CH4 and N2O fluxes from a wastewater treatment plant and nearby biogas production with discoveries of unexpected sources.* Environmental Research, 2022. **204**: p. 111978.
22. Delre, A., J. Mønster, and C. Scheutz, *Greenhouse gas emission quantification from wastewater treatment plants, using a tracer gas dispersion method.* Science of The Total Environment, 2017. **605-606**: p. 258-268.
23. Mikola, A., et al., *N2O emissions from secondary clarifiers and their contribution to the total emissions of the WWTP.* Water Science and Technology, 2014. **70**(4): p. 720-728.
24. Sun, S., et al., *Reduction and prediction of N2O emission from an Anoxic/Oxic wastewater treatment plant upon DO control and model simulation.* Bioresource Technology, 2017. **244**: p. 800-809.
25. Caniani, D., et al., *CO2 and N2O from water resource recovery facilities: Evaluation of emissions from biological treatment, settling, disinfection, and receiving water body.* Science of The Total Environment, 2019. **648**: p. 1130-1140.
26. Song, C., et al., *Oversimplification and misestimation of nitrous oxide emissions from wastewater treatment plants.* Nature Sustainability, 2024. **7**(10): p. 1348-1358.
27. Vasilaki, V., et al., *A decade of nitrous oxide (N(2)O) monitoring in full-scale wastewater treatment processes: A critical review.* Water Res, 2019. **161**: p. 392-412.
28. Zheng, H., K. Hanaki, and T. Matsuo, *Production of nitrous oxide gas during nitrification of wastewater.* Water Science and Technology, 1994. **30**(6): p. 133-141.
29. Li, P., et al., *The synergistic effects of dissolved oxygen and pH on N2O production in biological domestic wastewater treatment under nitrifying conditions.* Environmental Technology, 2015. **36**(13): p. 1623-1631.
30. Vasilaki, V., et al., *A knowledge discovery framework to predict the N2O emissions in the wastewater sector.* Water Research, 2020. **178**: p. 115799.
31. Peng, L., et al., *The effect of dissolved oxygen on N2O production by ammonia-oxidizing bacteria in an enriched nitrifying sludge.* Water Research, 2014. **66**: p. 12-21.
32. Colliver, B.B. and T. Stephenson, *Production of nitrogen oxide and dinitrogen oxide by autotrophic nitrifiers.* Biotechnology Advances, 2000. **18**(3): p. 219-232.
33. Law, Y., et al., *Nitrous oxide emissions from wastewater treatment processes.* Philosophical Transactions of the Royal Society B: Biological Sciences, 2012. **367**(1593): p. 1265-1277.
34. Ahn, J.H., T. Kwan, and K. Chandran, *Comparison of Partial and Full Nitrification Processes Applied for Treating High-Strength Nitrogen Wastewaters: Microbial Ecology through Nitrous Oxide Production.* Environmental Science & Technology, 2011. **45**(7): p. 2734-2740.

35. Peng, L., et al., *Smart operation of nitritation/denitritation virtually abolishes nitrous oxide emission during treatment of co-digested pig slurry centrate*. Water Research, 2017. **127**: p. 1-10.
36. Beaumont, H.J.E., et al., *Expression of nitrite reductase in Nitrosomonas europaea involves NsrR, a novel nitrite-sensitive transcription repressor*. Molecular Microbiology, 2004. **54**(1): p. 148-158.
37. Schulthess, R.v., M. Kühni, and W. Gujer, *Release of nitric and nitrous oxides from denitrifying activated sludge*. Water Research, 1995. **29**(1): p. 215-226.
38. Alinsafi, A., et al., *Nitrite effect on nitrous oxide emission from denitrifying activated sludge*. Process Biochemistry, 2008. **43**(6): p. 683-689.
39. Pan, Y., et al., *Electron competition among nitrogen oxides reduction during methanol-utilizing denitrification in wastewater treatment*. Water Research, 2013. **47**(10): p. 3273-3281.
40. Burgess, J.E., et al., *Dinitrogen oxide production by a mixed culture of nitrifying bacteria during ammonia shock loading and aeration failure*. Journal of Industrial Microbiology and Biotechnology, 2002. **29**(6): p. 309-313.
41. Butler, M.D., et al., *Nitrous oxide emissions for early warning of biological nitrification failure in activated sludge*. Water Research, 2009. **43**(5): p. 1265-1272.
42. Yu, R., et al., *Mechanisms and Specific Directionality of Autotrophic Nitrous Oxide and Nitric Oxide Generation during Transient Anoxia*. Environmental Science & Technology, 2010. **44**(4): p. 1313-1319.
43. Chandran, K., et al., *Nitrous oxide production by lithotrophic ammonia-oxidizing bacteria and implications for engineered nitrogen-removal systems*. Biochemical Society Transactions, 2011. **39**(6): p. 1832-1837.
44. Castro-Barros, C.M., et al., *Effect of aeration regime on N2O emission from partial nitritation-anammox in a full-scale granular sludge reactor*. Water Research, 2015. **68**: p. 793-803.
45. Schulthess, R.V. and W. Gujer, *Release of nitrous oxide (N2O) from denitrifying activated sludge: Verification and application of a mathematical model*. Water Research, 1996. **30**(3): p. 521-530.
46. Wunderlin, P., et al., *Mechanisms of N2O production in biological wastewater treatment under nitrifying and denitrifying conditions*. Water Research, 2012. **46**(4): p. 1027-1037.
47. Song, K., et al., *Impact of carbon sources on nitrous oxide emission and microbial community structure in an anoxic/oxic activated sludge system*. Clean Technologies and Environmental Policy, 2015. **17**(8): p. 2375-2385.
48. Adouani, N., et al., *Effect of the carbon source on N2O emissions during biological denitrification. Resources, Conservation and Recycling, 54(5), 299-302*. Resources, Conservation and Recycling, 2010. **54**: p. 299-302.
49. Zhang, X., et al., *Reduction of nitrous oxide emissions from partial nitrification process by using innovative carbon source (mannitol)*. Bioresource Technology, 2016. **218**: p. 789-795.
50. Hooper, A.B., et al., *Enzymology of the oxidation of ammonia to nitrite by bacteria*. Antonie van Leeuwenhoek, 1997. **71**(1): p. 59-67.

51. Schreiber, F., et al., *Nitric oxide and nitrous oxide turnover in natural and engineered microbial communities: biological pathways, chemical reactions, and novel technologies*. Frontiers in Microbiology, 2012. **Volume 3 - 2012**.
52. Su, Q., et al., *Abiotic Nitrous Oxide (N2O) Production Is Strongly pH Dependent, but Contributes Little to Overall N2O Emissions in Biological Nitrogen Removal Systems*. Environmental Science & Technology, 2019. **53**(7): p. 3508-3516.
53. Moffett, J.W., C.B. Tuit, and B.B. Ward, *Chelator-induced inhibition of copper metalloenzymes in denitrifying bacteria*. Limnology and Oceanography, 2012. **57**(1): p. 272-280.
54. Knowles, R., *Denitrification*. Microbiological Reviews, 1982. **46**(1): p. 43-70.
55. Schalk-Otte, S., et al., *Nitrous oxide (N2O) production by Alcaligenes faecalis during feast and famine regimes*. Water Research, 2000. **34**(7): p. 2080-2088.
56. Ahn, J.H., et al., *Spatial and Temporal Variability in Atmospheric Nitrous Oxide Generation and Emission from Full-Scale Biological Nitrogen Removal and Non-BNR Processes*. Water Environment Research, 2010. **82**(12): p. 2362-2372.
57. Hanaki, K., Z. Hong, and T. Matsuo, *Production of Nitrous Oxide Gas during Denitrification of Wastewater*. Water Science and Technology, 1992. **26**(5-6): p. 1027-1036.
58. Itokawa, H., K. Hanaki, and T. Matsuo, *Nitrous oxide production in high-loading biological nitrogen removal process under low cod/n ratio condition*. Water Research, 2001. **35**(3): p. 657-664.
59. Andalib, M., et al., *Correlation between nitrous oxide (N2O) emission and carbon to nitrogen (COD/N) ratio in denitrification process: a mitigation strategy to decrease greenhouse gas emission and cost of operation*. Water Science and Technology, 2017. **77**(2): p. 426-438.
60. Christensson, M., E. Lie, and T. Welander, *A comparison between ethanol and methanol as carbon sources for denitrification*. Water Science and Technology, 1994. **30**(6): p. 83-90.
61. Quan, X., et al., *Nitrous oxide emission and nutrient removal in aerobic granular sludge sequencing batch reactors*. Water Research, 2012. **46**(16): p. 4981-4990.
62. Amatya, I.M., et al., *Role of pH on biological Nitrification Process*. Journal of the Institute of Engineering, 2011. **8**(1-2): p. 119-125.
63. Gerardi, M.H., *Nitrification and Denitrification in the Activated Sludge Process*. 2002, New York: John Wiley and Sons, Inc.
64. Tchobanoglous;, G., et al., *Wastewater Engineering: Treatment and Resource Recovery*. 5 ed, ed. M.E. Inc. 2014: McGraw Hill.
65. Blum, J.-M., et al., *The pH dependency of N-converting enzymatic processes, pathways and microbes: effect on net N2O production*. Environmental Microbiology, 2018. **20**(5): p. 1623-1640.
66. Van Hulle, S.W.H., et al., *Engineering aspects and practical application of autotrophic nitrogen removal from nitrogen rich streams*. Chemical Engineering Journal, 2010. **162**(1): p. 1-20.
67. Suzuki, I., U. Dular, and S.C. Kwok, *Ammonia or Ammonium Ion as Substrate for Oxidation by Nitrosomonas europaea Cells and Extracts*. Journal of Bacteriology, 1974. **120**(1): p. 556-558.

68. Vadivelu, V.M., J. Keller, and Z. Yuan, *Effect of free ammonia on the respiration and growth processes of an enriched Nitrobacter culture*. Water Research, 2007. **41**(4): p. 826-834.
69. Anthonisen, A.C., et al., *Inhibition of nitrification by ammonia and nitrous acid*. Journal - Water Pollution Control Federation, 1976. **48**(5): p. 835-852.
70. Jiménez, E., et al., *Effect of pH and nitrite concentration on nitrite oxidation rate*. Bioresource Technology, 2011. **102**(19): p. 8741-8747.
71. Ruiz, G., D. Jeison, and R. Chamy, *Nitrification with high nitrite accumulation for the treatment of wastewater with high ammonia concentration*. Water Research, 2003. **37**(6): p. 1371-1377.
72. Thörn, M. and F. Sörensson, *Variation of nitrous oxide formation in the denitrification basin in a wastewater treatment plant with nitrogen removal*. Water Research, 1996. **30**(6): p. 1543-1547.
73. Pan, Y., et al., *Effect of pH on N2O reduction and accumulation during denitrification by methanol utilizing denitrifiers*. Water Research, 2012. **46**(15): p. 4832-4840.
74. Grunditz, C. and G. Dalhammar, *Development of nitrification inhibition assays using pure cultures of nitrosomonas and nitrobacter*. Water Research, 2001. **35**(2): p. 433-440.
75. Van Hulle, S.W.H., et al., *Influence of temperature and pH on the kinetics of the Sharon nitritation process*. Journal of Chemical Technology & Biotechnology, 2007. **82**(5): p. 471-480.
76. Hellinga, C., M.C.M. van Loosdrecht, and J.J. Heijnen, *Model Based Design of a Novel Process for Nitrogen Removal from Concentrated Flows*. Mathematical and Computer Modelling of Dynamical Systems, 1999. **5**(4): p. 351-371.
77. Boiocchi, R., K.V. Gernaey, and G. Sin, *A novel fuzzy-logic control strategy minimizing N2O emissions*. Water Research, 2017. **123**: p. 479-494.
78. Poh, L.S., et al., *N2O accumulation from denitrification under different temperatures*. Applied Microbiology and Biotechnology, 2015. **99**(21): p. 9215-9226.
79. Weiss, R.F. and B.A. Price, *Nitrous oxide solubility in water and seawater*. Marine Chemistry, 1980. **8**(4): p. 347-359.
80. Zumft, W.G., *Cell biology and molecular basis of denitrification*. Microbiology and Molecular Biology Reviews, 1997. **61**(4): p. 533-616.
81. Reino, C., et al., *Effect of temperature on N2O emissions from a highly enriched nitrifying granular sludge performing partial nitritation of a low-strength wastewater*. Chemosphere, 2017. **185**: p. 336-343.
82. Kosonen, H., et al., *Nitrous Oxide Production at a Fully Covered Wastewater Treatment Plant: Results of a Long-Term Online Monitoring Campaign*. Environmental Science & Technology, 2016. **50**(11): p. 5547-5554.
83. He, X., et al., *Quantifying greenhouse gas emissions from wastewater treatment plants: A critical review*. Environmental Science and Ecotechnology, 2025. **27**: p. 100606.
84. Sharma, A., et al., *Greenhouse gas emission potential of sewage treatment plants in Himachal Pradesh*. Scientific Reports, 2023. **13**(1): p. 9675.
85. Li, H., et al., *Methane and nitrous oxide emissions from municipal wastewater treatment plants in China: A plant-level and technology-specific study*. Environmental Science and Ecotechnology, 2024. **20**: p. 100345.

86. Santos, J.O., et al., *Greenhouse gas inventory of a state water and wastewater utility in Northeast Brazil*. Journal of Cleaner Production, 2015. **104**: p. 168-176.
87. Noyola, A., et al., *Methane correction factors for estimating emissions from aerobic wastewater treatment facilities based on field data in Mexico and on literature review*. Science of The Total Environment, 2018. **639**: p. 84-91.
88. Paredes, M.G., et al., *Methane emissions from anaerobic sludge digesters in Mexico: On-site determination vs. IPCC Tier 1 method*. Science of The Total Environment, 2019. **656**: p. 468-474.
89. Song, C., et al., *Methane Emissions from Municipal Wastewater Collection and Treatment Systems*. Environmental Science & Technology, 2023. **57**(6): p. 2248-2261.
90. Liu, Y., et al., *Methane emission from sewers*. Science of The Total Environment, 2015. **524-525**: p. 40-51.
91. Schneider, A.G., A. Townsend-Small, and D. Rosso, *Impact of direct greenhouse gas emissions on the carbon footprint of water reclamation processes employing nitrification–denitrification*. Science of The Total Environment, 2015. **505**: p. 1166-1173.
92. Delre, A., M. ten Hoeve, and C. Scheutz, *Site-specific carbon footprints of Scandinavian wastewater treatment plants, using the life cycle assessment approach*. Journal of Cleaner Production, 2019. **211**: p. 1001-1014.
93. Parravicini, V., et al., *Evaluation of greenhouse gas emissions from the European urban wastewater sector, and options for their reduction*. Science of The Total Environment, 2022. **838**: p. 156322.
94. El Abbadi, S.H., et al., *Benchmarking greenhouse gas emissions from US wastewater treatment for targeted reduction*. Nature Water, 2025. **3**(10): p. 1133-1143.
95. Chen, W., et al., *Understanding the greenhouse gas emissions from China's wastewater treatment plants: Based on life cycle assessment coupled with statistical data*. Ecotoxicology and Environmental Safety, 2023. **259**: p. 115007.
96. Borg, A., A.K. Dejardins, and H. Nersund, *Klimagassutslipp, veiledning for vannbransjen*. 2019: Hamar.
97. Ebie, Y., et al., *Development of Emissions Factor for the Decentralized Domestic Wastewater Treatment for the National Greenhouse Gas Inventory*. Journal of Water and Environment Technology, 2014. **12**(1): p. 33-41.
98. Boiocchi, R., et al. *Carbon Footprint and Energy Recovery Potential of Primary Wastewater Treatment in Decentralized Areas: A Critical Review on Septic and Imhoff Tanks*. Energies, 2023. **16**, DOI: 10.3390/en16247938.
99. Ploteau, M.-E.A., Anke; Nafo, Issa; Teichgräber, Burkhard, *Final report of the Phos4You partnership: deploying phosphorus recycling from wastewater in North-West Europe*, LIPPEVERBAND, Editor. 2021: Essen.
100. Boiocchi, R., et al., *Trends of N2O production during decentralized wastewater treatment: A critical review*. Journal of Environmental Chemical Engineering, 2025. **13**(1): p. 114627.
101. Ryals, R., et al., *Greenhouse gas fluxes from human waste management pathways in Haiti*. Journal of Cleaner Production, 2019. **226**: p. 106-113.
102. Silva Vinasco, J.P., *Greenhouse Gas Emissions from Ecotechnologies for Wastewater Treatment*. 1st ed. 2021: CRC Press. 194.

103. Ho, L., et al., *Spatial and temporal variations of greenhouse gas emissions from a waste stabilization pond: Effects of sludge distribution and accumulation.* Water Research, 2021. **193**: p. 116858.
104. Pang, J., et al., *How do hydraulic load and intermittent aeration affect pollutants removal and greenhouse gases emission in wastewater ecological soil infiltration systems?* Ecological Engineering, 2020. **146**: p. 105747.
105. Chen, J., et al., *Pollutants removal, greenhouse gases emission and functional genes in wastewater ecological soil infiltration systems: influences of influent surface organic loading and aeration mode.* Water Science and Technology, 2021. **83**(7): p. 1619-1632.
106. Pang, J., et al., *Does influent C/N ratio affect pollutant removal and greenhouse gas emission in wastewater ecological soil infiltration systems with/without intermittent aeration?* Water Science and Technology, 2020. **81**(4): p. 668-678.
107. Zheng, F., et al., *Nitrogen removal and N2O emission by shunt distributing wastewater in aerated or non-aerated subsurface wastewater infiltration systems under different shunt ratios.* Water Science and Technology, 2018. **78**(2): p. 329-338.
108. Maucieri, C., et al., *A review on the main affecting factors of greenhouse gases emission in constructed wetlands.* Agricultural and Forest Meteorology, 2017. **236**: p. 175-193.
109. Domingo-Félez, C., et al., *Variability and Uncertainty Analysis of N2O Emissions from WWTP to Improve the Accuracy of Emission Factors and the Design of Monitoring Strategies.* ACS ES&T Water, 2024. **4**(6): p. 2542-2552.
110. Schouten, P., et al., *Evaluation of Fugitive Greenhouse Gas Emissions from Decentralized Wastewater Treatment Plants*, in *World Environmental and Water Resources Congress 2013*. 2013. p. 3050-3061.
111. Daelman, M.R.J., et al., *Methane emission during municipal wastewater treatment.* Water Research, 2012. **46**(11): p. 3657-3670.
112. Fredenslund, A.M., et al., *The Danish national effort to minimise methane emissions from biogas plants.* Waste Management, 2023. **157**: p. 321-329.
113. Gålfalk, M. and D. Bastviken, *In Situ Observations Reveal Underestimated Greenhouse Gas Emissions from Wastewater Treatment with Anaerobic Digestion – Sludge Was a Major Source for Both CH4 and N2O.* Environmental Science & Technology, 2025. **59**(34): p. 18146-18155.
114. Oshita, K., et al., *Methane and nitrous oxide emissions following anaerobic digestion of sludge in Japanese sewage treatment facilities.* Bioresource Technology, 2014. **171**: p. 175-181.
115. Brown, S., N. Beecher, and A. Carpenter, *Calculator Tool for Determining Greenhouse Gas Emissions for Biosolids Processing and End Use.* Environmental Science & Technology, 2010. **44**(24): p. 9509-9515.
116. Miller-Robbie, L., et al., *Life cycle energy and greenhouse gas assessment of the co-production of biosolids and biochar for land application.* Journal of Cleaner Production, 2015. **91**: p. 118-127.
117. Garrido-Baserba, M., et al., *Selecting sewage sludge treatment alternatives in modern wastewater treatment plants using environmental decision support systems.* Journal of Cleaner Production, 2015. **107**: p. 410-419.

118. Piippo, S., M. Lauronen, and H. Postila, *Greenhouse gas emissions from different sewage sludge treatment methods in north*. Journal of Cleaner Production, 2018. **177**: p. 483-492.
119. Yoshida, H., et al., *Life cycle assessment of sewage sludge management options including long-term impacts after land application*. Journal of Cleaner Production, 2018. **174**: p. 538-547.
120. Parravicini, V.F., Ahlem; Delre, Antonio; Gutierrez, Oriol; Duan, Haoran, *Full-scale quantification of N2O and CH4 emissions from urban water systems*, in *Quantification and Modelling of Fugitive Greenhouse Gas Emissions from Urban Water Systems*, L. Ye, J. Porro, and I. Nopens, Editors. 2022, IWA Publishing. p. 91-131.
121. Vangsgaard, A.K. and J.A. Madsen, *MUDP Lattergaspulje. Dataopsamling på måling og reduktion af lattergasemissioner fra renseanlæg*. 2020, Danish EPA.
122. Abdolrezayi, A., et al., *Phosphorus recovery from sewage sludge ash: life cycle inventory and critical review of LCA case studies*. Journal of Environmental Management, 2025. **389**: p. 125620.
123. Pap, S.T., Mark; Bremner, Barbara, *Demonstrator I5: P adsorption for small scale use (FILTRAFLO™-P with CCM adsorbent)*, in *Technical report of the Phos4You partnershipon processes to recover phosphorus from wastewater*, M.-E.A. Ploteau, Anke; Nafo, Issa; Teichgräber, Burkhard, Editor. 2021: Essen. p. 186-210.
124. Escudero, A.R., Lena-Dorothea; Pahl, Ole, *Demonstrator I4: Microalgae to recover P from small-scale WWTPs.*, in *Technical report of the Phos4You partnershipon processes to recover phosphorus from wastewater*, M.-E.A. Ploteau, Anke; Nafo, Issa; Teichgräber, Burkhard, Editor. 2021: Essen. p. 161-185.
125. Moore, D.P., et al., *Underestimation of Sector-Wide Methane Emissions from United States Wastewater Treatment*. Environmental Science & Technology, 2023. **57**(10): p. 4082-4090.
126. STOWA, *Risico-inschatting emissie lachgas vanuit Nederlandse rioolwaterzuiveringen. Resultaten meetonderoek voor verificatie eenvoudige risico-inschatting*. 2019, STOWA.
127. Wang, D., et al., *Greenhouse gas emissions from municipal wastewater treatment facilities in China from 2006 to 2019*. Scientific Data, 2022. **9**(1): p. 317.
128. Morales-Rico, P., et al., *A simplified open flux chamber method for the measurement of greenhouse gas emissions from activated sludge reactors*. Journal of Water and Climate Change, 2024. **15**(5): p. 2127-2140.
129. Sieranen, M., et al., *Seasonality of nitrous oxide emissions at six full-scale wastewater treatment plants*. Water Science and Technology, 2023. **89**(3): p. 603-612.
130. NIRAS, *Lattergas fra renseanlæg - Foranalyse vedrørende regulering*. 2022.
131. Johansson, C.Y., Johan ; Gålfalk, Magnus; Sehlén, Robert ; Påledal, Sören Nilsson, *Innovativ teknik för mätning av växthusgaser från avloppsreningsverk*. 2020.
132. Ledermann, L.D., et al., *AWAIRE: Development and testing of techniques for measuring N2O emissions from wastewater treatment plants (Revised)*. 2025.
133. Koutsou, O.P., G. Gatidou, and A.S. Stasinakis, *Domestic wastewater management in Greece: Greenhouse gas emissions estimation at country scale*. Journal of Cleaner Production, 2018. **188**: p. 851-859.

134. Marinelli, E., et al., *Validation of an evidence-based methodology to support regional carbon footprint assessment and decarbonisation of wastewater treatment service in Italy.* Water Research, 2021. **207**: p. 117831.
135. Yang, M., et al., *Greenhouse gas emissions from wastewater treatment plants in China: Historical emissions and future mitigation potentials.* Resources, Conservation and Recycling, 2023. **190**: p. 106794.
136. WHO, *Country files for SDG 6.3.1 Proportion of wastewater safely treated*. 2025.
137. *The Global Methane Pledge*. [cited 2025 30 October]; Available from: https://www.globalmethanepledge.org/.
138. *Member States' emission targets: About Effort Sharing*. [cited 2025 30 October]; Available from: https://climate.ec.europa.eu/eu-action/effort-sharing-member-states-emission-targets/about-effort-sharing_en.
139. *Effort sharing 2021-2030: targets and flexibilities*. [cited 2025 30 October]; Available from: https://climate.ec.europa.eu/eu-action/effort-sharing-member-states-emission-targets/effort-sharing-2021-2030-targets-and-flexibilities_en.
140. *Klimaplan for en grøn affaldssektor og cirkulær økonomi*. 2020, Klima-, Energi- og Forsyningsministeriet.
141. Bayley, C.C., et al., *Proposals for regulatory methods to reduce nitrous oxide emissions from treatment facilities*. 2025: Odense. p. 154.
142. Never, B., *Wastewater systems and energy saving in urban India*, in *Governing the Water-Energy-Food Nexus*. 2016, Deutsches Institut für Entwicklungspolitik: Bonn.
143. *Project Profile: McAlpine Creek Wastewater Management Facility*. 2018 [cited 2025 July 1]; Available from: https://chptap.ornl.gov/profile/126/McAlpineWWTP-Project_Profile.pdf.
144. Porto, P.M., et al., *Biogas energy recovery in brazilian wastewater treatment plants: A multi-level perspective on technological transition.* Energy Reports, 2025. **13**: p. 4691-4704.
145. SABESP. *Inovação e Tecnologia - Biogás*. 2023; Available from: https://www.sabesp.com.br/a-sabesp/inovacao-e-tecnologia/biogas.
146. SANEPAR. *Relatório Integrado*. 2023; Available from: https://ri.sanepar.com.br/docs/Sanepar-2023-12-31-n8Wb9Gjn.pdf.
147. Abiogás, *PNBB - Programa Nacional do Biogás e Biometano*. 2022, Associação Brasileira do Biogás.
148. Hao, X., et al., *Resource Recovery from Wastewater: What, Why, and Where?* Environmental Science & Technology, 2024. **58**(32): p. 14065-14067.
149. Hao, X., et al., *Energy recovery from wastewater: Heat over organics.* Water Research, 2019. **161**: p. 74-77.
150. *Vienna's large-scale heat pumps providing clean energy*. 2025 [cited 2025 July 1]; Available from: https://positionen.wienenergie.at/en/projects/viennas-large-scale-heat-pumps/.
151. *Utilising sewage waste-water heat in district heating systems, Serbia*, in *European Commission DG Environment News Alert Service*. 2022, SCU, The University of the West of England: Bristol.
152. Liu, R., et al., *Contribution of solar photovoltaic to the decarbonization of wastewater treatment plants in China.* Desalination, 2025. **611**: p. 118935.

153. Akgül Tözöm, S., *Assessment of the role of photovoltaic systems in reducing the carbon footprint of wastewtaer treatment plants.* Global NEST Journal, 2023. **25**(8): p. 139-145.
154. Johansson, M., *Urine Separation: Closing the Nutrient Cycle*. 2000: Stockholm, Sweden.
155. Badeti, U., et al., *Impact of source-separation of urine on effluent quality, energy consumption and greenhouse gas emissions of a decentralized wastewater treatment plant.* Process Safety and Environmental Protection, 2021. **150**: p. 298-304.
156. Maurer, M., W. Pronk, and T.A. Larsen, *Treatment processes for source-separated urine.* Water Research, 2006. **40**(17): p. 3151-3166.
157. Yang, W., J. Li, and X. Yang, *Features and Applications of Urine Stabilization Methods: A Review.* Frontiers in Sustainability, 2021. **Volume 2 - 2021**.
158. *Valorisation of Urine Nutrients in Africa (VUNA Final Report): Promoting Sanitation & Nutrient Recovery through Urine Separation*, B. Etter, K.M. Udert, and T. Gounden, Editors. 2015: Dübendorf, Switzerland.
159. Espelund, G. *eThekwini chose waterless toilets*. Stockholm Water Front, 2018. **Decmber**, 14-17.
160. Aliahmad, A., W. Kanda, and J. McConville, *Urine recycling - Diffusion barriers and upscaling potential; case studies from Sweden and Switzerland.* Journal of Cleaner Production, 2023. **414**: p. 137583.
161. Johansson, M., E. Kvarnström, and A.R. Stintzing *Going to Scale with Urine Diversion in Sweden from Individual Households to Municipal Systems in 15 Years*. 2009.
162. GWOPA, *Towards a Sanitation Strong GWOPA Strategy*. 2023.
163. Ouedraogo, N.I., et al. *Characterization and Methanogenic Potential Evaluation of Faecal Sludge: Case of the Kossodo Biogas Plant in Ouagadougou*. Sustainability, 2023. **15**, 16401 DOI: 10.3390/su152316401.
164. Acuña, V., et al., *Development of a decision-support system to select nature-based solutions for domestic wastewater treatment.* Blue-Green Systems, 2023. **5**(2): p. 235-251.
165. UNEP, *Wastewater - Turning problem to solution*, in *UNEP Rapid Response Assessment*. 2023: Nairobi
166. *Paques biomaterials creates truly biodegradable bioplastic from wastewater.* Groene Chemie Nieuwe Economie, 2024.
167. Zahra, S.A., et al., *Rethinking characterization, application, and importance of extracellular polymeric substances in water technologies.* Current Opinion in Biotechnology, 2024. **89**: p. 103192.
168. Mitura, S., A.A.-O. Sionkowska, and A. Jaiswal, *Biopolymers for hydrogels in cosmetics: review.* Journal of Materials Science Materials in medicine, 2020. **31**(6).
169. *Materials of the future extracted from wastewater*, in *Aquatech*. 2024.
170. Fang, L.L., et al., *Life cycle assessment as development and decision support tool for wastewater resource recovery technology.* Water Research, 2016. **88**: p. 538-549.
171. Styles, D., et al., *Life Cycle Assessment of Biofertilizer Production and Use Compared with Conventional Liquid Digestate Management.* Environmental Science & Technology, 2018. **52**(13): p. 7468-7476.
172. Noyola, A., et al., *Reduction of Greenhouse Gas Emissions From Municipal Wastewater Treatment in Mexico Based on Technology Selection.* CLEAN – Soil, Air, Water, 2016. **44**(9): p. 1091-1098.

173. Duan, H., et al., *Insights into Nitrous Oxide Mitigation Strategies in Wastewater Treatment and Challenges for Wider Implementation*. Environmental Science & Technology, 2021. **55**(11): p. 7208-7224.

174. Tallec, G., et al., *Nitrous oxide emissions from secondary activated sludge in nitrifying conditions of urban wastewater treatment plants: Effect of oxygenation level*. Water Research, 2006. **40**(15): p. 2972-2980.

175. Massara, T.M., et al., *A review on nitrous oxide (N2O) emissions during biological nutrient removal from municipal wastewater and sludge reject water*. Science of The Total Environment, 2017. **596-597**: p. 106-123.

176. Duan, H., et al., *Mitigating nitrous oxide emissions at a full-scale wastewater treatment plant*. Water Research, 2020. **185**: p. 116196.

177. Ni, B.-J., et al., *Full-Scale Modeling Explaining Large Spatial Variations of Nitrous Oxide Fluxes in a Step-Feed Plug-Flow Wastewater Treatment Reactor*. Environmental Science & Technology, 2015. **49**(15): p. 9176-9184.

178. Li, H., et al., *N2O generation and emission from two biological nitrogen removal plants in China*. Desalination and Water Treatment, 2016. **57**(25): p. 11800-11806.

179. Willis, J.L., et al., *A greenhouse gas source of surprising significance: anthropogenic CO(2) emissions from use of methanol in sewage treatment*. Water Sci. Technol., 2017. **75**(9-10): p. 1997-2012.