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# Operational Greenhouse Gas Emissions of Public Wastewater Treatment Plants in Korea Estimated from National Sewerage Statistics

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**Abstract**

Public wastewater treatment plants (WWTPs) emit greenhouse gases directly from treatment and indirectly through electricity use. Resource recovery can offset part of these emissions. This study estimated the operational greenhouse gas emissions of 610 public WWTPs in Korea (≥500 $m^3$/d; 96.2% of the national treated flow). We used the 2024 Sewerage Statistics, IPCC Tier 1 methods and AR6 $GWP_{100}$ values. Annual gross emissions were 5.207 Mt $CO_2$-eq, and the flow-weighted intensity was 0.723 kg $CO_2$-eq/$m^3$. On this basis, $N_2O$ from treatment and receiving water was the largest source (43.5%), followed by electricity (31.8%). The arithmetic plant mean was higher, at 1.039 kg $CO_2$-eq/$m^3$, and electricity was its largest component (53.2%). Small plants are numerous and use more electricity per cubic meter, and this explains almost all of the difference between the two measures. Intensity fell from 1.137 to 0.720 kg $CO_2$-eq/$m^3$ between the smallest and largest capacity classes. However, plants of ≥100,000 $m^3$/d produced 67.3% of the annual emissions. Conditional credits for effluent reuse, biogas use and sludge recovery amounted to 0.457 Mt $CO_2$-eq (8.8%). At plants that produced biogas, $CH_4$ from surplus biogas exceeded the biogas credit when the surplus was assumed to be vented. The results depend strongly on the $N_2O$ emission factor for treatment. Over its IPCC range, the mean net intensity varied from 0.680 to 1.559 kg $CO_2$-eq/$m^3$.

**Keywords:** global warming potential; greenhouse gas emissions; environmental credit; wastewater treatment plant; IPCC guidelines; anaerobic digestion; sludge management; Korea

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## 1. Introduction

Reducing greenhouse gas (GHG) emissions is essential for limiting climate change and for sustainable development [1]. Korea compiles a national GHG inventory to estimate and manage emissions by sector [2]. However, meeting the national reduction target requires more than the emission totals: it requires quantitative evidence on which sources dominate at each facility and where management effort should be concentrated. Mitigation strategies therefore need to identify the composition and location of emission sources as well as their magnitude.

Wastewater treatment plants (WWTPs) are a representative case. They are public infrastructure for domestic sewage treatment and water quality management, but they are also energy-intensive facilities that consume large amounts of electricity for influent pumping, aeration, and sludge handling [3]. In 2024, public WWTPs in Korea consumed about 3.6 TWh of electricity [4], roughly 0.6% of the national generation of 595.6 TWh in

the same year [5]. This consumption appears as indirect emissions (Scope 2) from power generation [6].

Treatment processes also emit GHGs directly (Scope 1). Carbon dioxide ($CO_2$), methane ($CH_4$), and nitrous oxide ($N_2O$) are the direct emissions reported for WWTPs [7]. $CO_2$ is released when organic matter is oxidized under aerobic conditions, but because it is mostly biogenic carbon within the short-term carbon cycle, it is excluded from emission totals [8,9]. $CH_4$ forms where organic matter degrades anaerobically, as in digesters and sludge storage tanks [10], and $N_2O$ is produced during nitrification and denitrification in biological nitrogen removal [11]. Direct and indirect emissions thus arise from different mechanisms at the same plant, and both must be evaluated together to describe the actual emission structure.

The national GHG inventory alone cannot reveal the integrated emission structure of WWTPs. Emissions are estimated from sectoral aggregates, so differences arising from the treatment process and operating conditions of individual plants are not visible. Electricity-related emissions are assigned to the energy sector, which obscures the balance between direct and indirect emissions at the facility level, and emissions avoided through resource recovery are not counted at all. The current system is therefore useful for tracking the sector total but offers little of the information needed to rank emission sources and management priorities by facility type.

Life cycle assessment (LCA) studies of individual WWTPs have been conducted to fill this gap. Li et al. analyzed the operational energy use and GHG emissions of WWTPs in Seoul and identified electricity consumption as the main contributor [12]. Szulc et al., Bhowmik et al., and Tsangas et al. applied LCA to various plants and showed that treatment process, operating conditions, and energy use govern the environmental impact [13–15]. These studies describe the emission structure of single plants in detail, but their results cannot readily be generalized to the national level. WWTP emissions depend on the treatment process, plant capacity, nitrogen removal, sludge disposal route, and the presence of anaerobic digestion [16], and these attributes vary widely among Korean public WWTPs.

WWTPs are also being redefined as resource recovery facilities that recover energy and water rather than merely removing pollutants [17]. Biogas from anaerobic digestion, effluent reuse, and sludge valorization replace fossil fuels, tap water, and raw materials and can thereby reduce GHG emissions [18]. At the national scale, El Abbadi et al. compiled an emission inventory of 15,863 WWTPs in the United States [19]. Chen et al. estimated the emissions of 247 WWTPs in China and reported a mean intensity of 0.45 kg $CO_2$-eq/$m^3$ [20]. In Korea, plant-level assessments have been regional in scope, such as that of the WWTPs in Seoul [12]. Field measurements of $CH_4$ and $N_2O$ have also been reported for Korean plants [21,22], but they cover a small number of plants and processes. The measured $N_2O$ emission factors also differ widely. One study reported 0.94 to 26.73 g $N_2O$/kg TN across four processes [23]. These studies provide process-level evidence. However, they do not show how emissions are distributed among the public WWTPs of the country, and the national inventory reports only sectoral totals. This study addresses this gap in three respects. First, we apply a single accounting framework to 610 public WWTPs taken from the national statistics. These plants treat 96.2% of the national effluent. Second, we report both the arithmetic plant mean and the flow-weighted intensity. Small plants dominate by number and large plants dominate by flow, so the two measures rank the emission sources differently. Third, we report the credits from resource recovery separately from the gross emissions, as conditional values. In Korea, where the uptake of resource recovery facilities differs markedly from plant to plant, the comparison of emission characteristics among plants can change depending on whether such credits are included.

This study therefore quantified the operational GHG emissions of public WWTPs in Korea from the Sewerage Statistics of the Ministry of Environment. Specifically, we (i) estimated the direct process emissions (Scope 1) and the electricity-related indirect emissions (Scope 2) of all qualifying plants on a common functional unit; (ii) quantified the environmental credits from effluent reuse, biogas use, and sludge valorization and derived the net emissions; and (iii) analyzed the emission characteristics and the contribution of each source by treatment process and plant capacity. The results are intended to provide a quantitative basis for GHG mitigation and carbon neutrality policies in the Korean wastewater sector.

## 2. Materials and Methods

### *2.1. System Boundary and Functional Unit*

The system boundary was set to the operational stage of public WWTPs. It covers the direct emissions from wastewater and sludge treatment, the indirect emissions from electricity use, and the environmental credits from biogas use, effluent reuse, and sludge valorization (Figure 1). The functional unit is 1 $m^3$ of treated effluent. We divided the annual emissions and credits of each plant by its annual effluent volume and expressed them in kg $CO_2$-eq/$m^3$. The normalized values were then compared by treatment process and plant capacity. The direct emissions grouped as Scope 1 in this study include $N_2O$ in the receiving water and the treatment of outsourced sludge. These emissions occur outside the plant and are not Scope 1 emissions of the operator under the GHG Protocol [6]. Section 3.4 shows them separately. The credits are avoided emissions and are not deducted from the Scope 1 or Scope 2 totals.

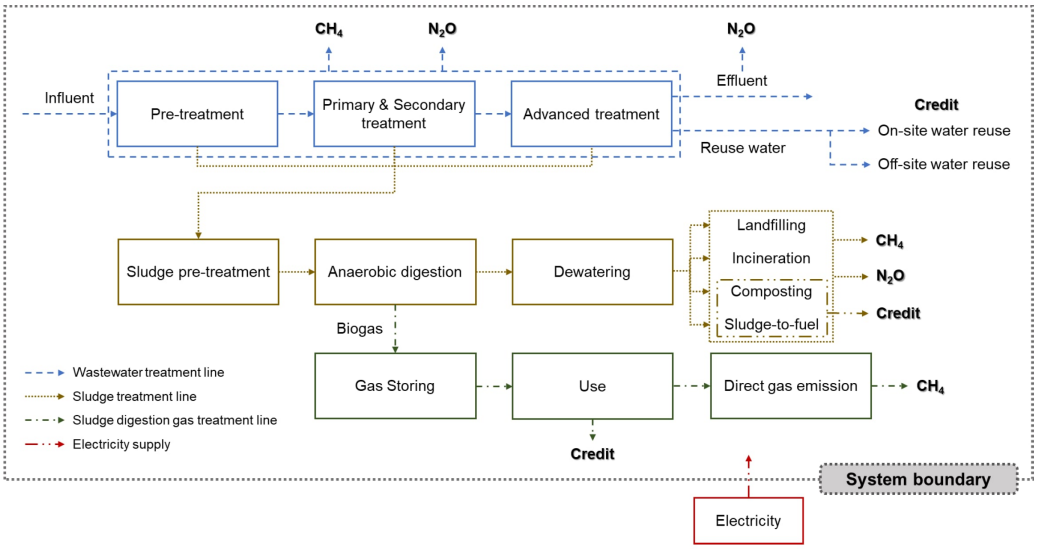


**Figure 1.** System boundary of the operational-stage assessment of public wastewater treatment plants. The electricity arrow denotes the supply to all processes within the system boundary (Scope 2).

### *2.2. Data Sources and Plant Selection*

The 2024 Sewerage Statistics of the Ministry of Environment were the primary data source [4]. Four datasets—plant inventory, sludge generation and treatment, digester operation, and effluent reuse—were merged into a single analysis dataset; the variables taken from each dataset and their use in the accounting are listed in Table 1. Plants with a design capacity of at least 500 $m^3$/d were selected, following the facility classification in the Guidelines for Establishing Comprehensive Sewerage Plans [24]; 724 of the 4469 public WWTPs in the 2024 statistics met this criterion. Of these, 101 plants with a daily inflow

below 500 $m^3$/d were excluded, leaving 623 plants. A further 13 plants belonging to process combinations with fewer than five plants were excluded as unrepresentative for group comparison, giving a final set of 610 plants (Figure 2). These plants account for 96.2% of the national public WWTP effluent and 92.4% of the electricity consumption in 2024. The 13 plants were excluded from all calculations; they account for 2.5% of the effluent and 2.3% of the electricity use of the 623 plants. The treatment processes reported in the statistics were grouped into five families—activated sludge (AS), biological nutrient removal (BNR), sequencing batch reactor (SBR), membrane bioreactor (MBR), and biofilm—and plants reporting processes from more than one family were classified as combined processes (Supplementary Table S1). Plant capacity was divided into four classes following Oh et al.: 500 to <10,000, 10,000 to <50,000, 50,000 to <100,000, and ≥100,000 $m^3$/d [25].

**Table 1.** Summary of the datasets used for greenhouse gas accounting.

| Dataset | Parameters | GHG Accounting Items |
| --- | --- | --- |
| Public WWTP inventory | Influent and effluent T-N and BOD concentrations, daily inflow and effluent volumes, design capacity, electricity consumption, treatment process | Scope 1: direct $CH_4$ and $N_2O$ emissions from biological treatment; Scope 2: indirect emissions from electricity use |
| Sludge generation and treatment | Sludge mass by treatment method, moisture content | GHG emissions by sludge disposal route; avoided-emission credits from sludge recovery |
| Digester operation | Operating days, surplus biogas volume, biogas allocation by use | Scope 1: direct $CH_4$ emissions; avoided-emission credits from biogas use |
| Effluent reuse | On-site reuse volume (washing, cooling, cleaning, etc.), off-site reuse volume (cleaning, agriculture, industry, etc.) | Avoided-emission credits for the water source substituted by each use |

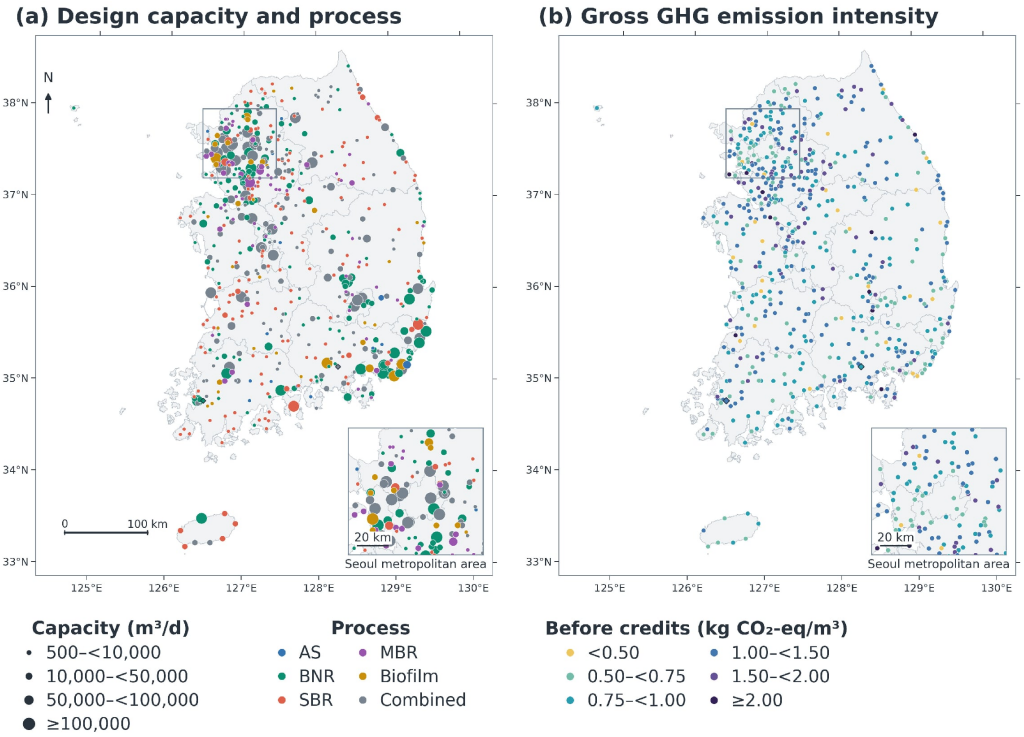


**Figure 2.** Spatial distribution of the 610 public wastewater treatment plants by (**a**) design capacity and treatment process and (**b**) gross greenhouse gas emission intensity before credits. Insets show the Seoul metropolitan area. Tick labels give longitude and latitude. The maps are drawn in Korea 2000/Unified CS (EPSG:5179).

The reporting status of each variable and the treatment of missing values are summarized in Supplementary Table S2. Influent and effluent quality, flows, and electricity consumption were reported by all 610 plants. The reported values were used without outlier screening, except for the sludge moisture content (Table S2). Where a sludge disposal route, digester record, or reuse record was not reported, we set the value to zero for calculation. An unreported activity is not necessarily absent. Intermediately treated sludge whose final destination could not be identified was excluded from the assessment. This sludge went to drying, solidification, carbonization, product feedstock, and landfill cover material. It amounted to 1.96 million t/year, or about 45% of the reported generation of 4.34 million t/year on a dewatered basis. For the 120 plants whose sludge moisture content was missing or exceeded 100%, the median of the valid reports (81.35%) was applied.

### *2.3. Greenhouse Gas Accounting*

GHG emissions were estimated primarily with the Tier 1 methods of the 2019 Refinement to the IPCC Guidelines; items not revised in 2019 followed the 2006 Guidelines, and items covered by neither were taken from IPCC-related reports, previous studies, and Korean emission factors. Emissions were converted to kg $CO_2$-eq with the 100-year global warming potential (GWP) values of the IPCC Sixth Assessment Report (AR6; $CO_2 = 1$, $CH_4 = 27$, $N_2O = 273$) and normalized by the effluent volume [26]. The sum of Scope 1 and Scope 2 before credits is defined as the gross emission intensity, and the net emission intensity after subtracting the credits was calculated with Equation (1). Values by treatment process and capacity class are the arithmetic means of the plant-level values in each group.

$$E_{net} = E_{scope1} + E_{scope2} - Credit_{resource} - Credit_{water} \quad (1)$$

where $E_{scope1}$ is the direct emission from wastewater and sludge treatment and from the venting of surplus digester gas, and $E_{scope2}$ is the indirect emission from electricity use [6]. $Credit_{resource}$ is the avoided emission from substituting grid electricity and natural gas with biogas and from substituting fertilizer and fossil fuel with recovered sludge (composting and fuel production), and $Credit_{water}$ is the avoided emission from substituting tap water and industrial water with reused effluent. The factors applied and their sources are summarized in Section 2.3.5; only the variables of each equation are described in the text.

#### 2.3.1. Direct $CH_4$ and $N_2O$ Emissions from Wastewater Treatment (Scope 1)

$CH_4$ emissions from biological treatment were calculated from the BOD removed, taken as the difference between the influent and effluent BOD mass loads, with the $B_0$ and MCF of the IPCC (2019) wastewater chapter [27] (Equation (2)). This is a simplified form of the IPCC Tier 1 equation. The IPCC equation subtracts the organic matter removed as sludge from the organic load and estimates $CH_4$ from the discharged effluent separately. We did not subtract the sludge term, because the statistics report the dewatered sludge mass but not the dry mass of raw sludge or its BOD content. We also did not estimate $CH_4$ from the effluent. The organic matter transferred to sludge is therefore included in Equation (2), while the sludge-related emissions are calculated separately from the sludge mass (Section 2.3.3). The two terms overlap in part, so the treatment $CH_4$ is an upper estimate. $N_2O$ emissions were calculated from the nitrogen entering biological treatment and the nitrogen discharged with the effluent, as in Equations (2)–(4) [27]. Equation (4) is the IPCC estimate of the $N_2O$ formed in the receiving water from the nitrogen discharged with the effluent. This emission does not occur within the plant, but it was included because it is determined by the nitrogen remaining after treatment. The two $N_2O$ terms refer to emissions at different places, during treatment and after discharge. We added them as in

the IPCC method. They are not derived from a nitrogen mass balance. We did not assume equal influent and effluent flows, because the reported values differ at 588 of the 610 plants.

$$CH_{4,treat}\ (kg\ CH_4/d) = (C_{in,BOD} \times Q_{in} - C_{eff,BOD} \times Q_{eff}) \times B_0 \times MCF_j \times 10^{-3} \quad (2)$$

$$N_2O_{treat}\ (kg\ N_2O/d) = C_{in,T\text{-}N} \times Q_{in} \times EF_{treat} \times 44/28 \times 10^{-3} \quad (3)$$

$$N_2O_{effluent}\ (kg\ N_2O/d) = C_{eff,T\text{-}N} \times Q_{eff} \times EF_{effluent} \times 44/28 \times 10^{-3} \quad (4)$$

where $C_{in,BOD}$ and $C_{eff,BOD}$ are the influent and effluent BOD concentrations and $C_{in,T\text{-}N}$ and $C_{eff,T\text{-}N}$ are the influent and effluent T-N concentrations (mg/L); $Q_{in}$ and $Q_{eff}$ are the daily influent and effluent volumes ($m^3/d$); $B_0$ is the maximum $CH_4$ producing capacity; $MCF_j$ is the methane correction factor representing the degree of anaerobic conditions in the treatment process; $EF_{treat}$ and $EF_{effluent}$ are the fractions of nitrogen converted to $N_2O$-N in the treatment process and in the receiving water, respectively; $10^{-3}$ converts mg/L to $kg/m^3$; and 44/28 converts $N_2O$-N to $N_2O$.

The same emission factors were applied to all plants. All the plants analyzed are subject to the effluent T-N standard of 20 mg/L [28]. They remove nitrogen in practice, whatever the reported process name (Table S8). The $N_2O$ emission factor, however, differs among processes. Even for the same process, it changes with location, operation and influent characteristics. Korean field measurements have mostly reported values lower than the IPCC default used here (25.14 g $N_2O$/kg influent N) [21–23], although one process in [23] showed a higher value (26.73 g $N_2O$/kg TN). The differences in $N_2O$ among the process groups in this study therefore reflect influent nitrogen loads only. We do not interpret them as process effects. Plant-level reported $N_2O$ emissions are not publicly available in Korea, so we could not validate the estimates against reported values.

#### 2.3.2. Direct $CH_4$ Emissions from Surplus Digester Gas (Scope 1)

Surplus digester gas that is not used for power generation or heating may be flared or vented. Because plant-level flaring records were unavailable, all surplus biogas was assumed to be vented, and the $CH_4$ emission was estimated conservatively with the UNFCCC equation (Equation (5)) [29].

$$CH_{4,dig}\ (kg\ CH_4/d) = V_{excess} \times f_{CH_4} \times \rho_{CH_4} \quad (5)$$

where $V_{excess}$ is the daily surplus biogas volume ($m^3/d$), and $f_{CH_4}$ and $\rho_{CH_4}$ are the $CH_4$ volume fraction and density of biogas. Plants without digesters were set to zero. Biogas used in engines or boilers was not counted as a direct emission but only as a credit (Section 2.3.5). Leakage and unburned methane from engines or boilers are distinct from surplus venting. They are not measured here. Table S7 compares flare operation and illustrative losses of 1% and 3% of recorded methane production.

#### 2.3.3. Emissions from Sludge Treatment (Scope 1 and Scope 2)

Emissions from final sludge disposal were calculated for four routes—incineration, landfilling, composting, and fuel production—with Equations (6)–(12). The sludge mass of each route is the sum of on-site and outsourced treatment, and where the factors are expressed on a dry basis, the mass was converted with the moisture content. For incineration, only $CH_4$ and $N_2O$ were calculated following the Tier 1 method of the IPCC (2006) waste incineration chapter [30], and biogenic $CO_2$ was excluded [30]. Landfilling followed the simplified Tier 1 method of the IPCC (2000) Good Practice Guidance [31,32]. Composting followed the Tier 1 method of the IPCC (2006) biological treatment chapter [33] with the dry-basis emission factors. The wet-basis factors assume a moisture content of 60%, which is lower than that of the sludge in this study [33]. For fuel production, the natural gas and

electricity consumed per unit of dry sludge were multiplied by the natural gas combustion and grid emission factors and assigned to Scope 1 and Scope 2, respectively [34–36]. These literature values do not cover the full heat required to dry dewatered sludge, so the emission of SRF production may be underestimated. The fertilizer and fossil fuel substitution achieved by sludge recovery was credited as described in Section 2.3.5. The energy included therefore differs among the routes. The SRF factor includes the natural gas and electricity used for fuel production. The incineration and composting factors cover only the process emissions of $CH_4$ and $N_2O$, because the statistics do not report the auxiliary fuel and electricity of these routes.

$$CH_{4,inc}\ (kg/d) = M_{inc} \times EF_{CH_4} \times 10^{-3} \quad (6)$$

$$N_2O_{inc}\ (kg/d) = M_{inc} \times EF_{N_2O} \times 10^{-3} \quad (7)$$

$$CH_{4,lan}\ (kg/d) = M_{lan} \times MCF \times DOC \times DOC_f \times F \times 16/12 \times (1 - OX) \quad (8)$$

$$CH_{4,com}\ (kg/d) = M_{com} \times (1 - MC/100) \times EF_{CH_4,com} \quad (9)$$

$$N_2O_{com}\ (kg/d) = M_{com} \times (1 - MC/100) \times EF_{N_2O,com} \quad (10)$$

$$CO_{2,SRF,scope\ 1}\ (kg\ CO_2\text{-}eq/d) = M_{SRF} \times (1 - MC/100) \times C_{NG} \times EF_{NG} \times 10^{-6} \quad (11)$$

$$CO_{2,SRF,scope\ 2}\ (kg\ CO_2\text{-}eq/d) = M_{SRF} \times (1 - MC/100) \times C_{grid} \times EF_{grid} \quad (12)$$

where $M_{inc}$, $M_{lan}$, $M_{com}$, and $M_{SRF}$ are the daily sludge masses sent to each route (t/d), and MC is the moisture content (%). $EF_{CH_4}$ and $EF_{N_2O}$ and $EF_{CH_4,com}$ and $EF_{N_2O,com}$ are the $CH_4$ and $N_2O$ emission factors for incineration and composting; MCF, DOC, $DOC_f$, F, and OX are the methane correction factor, degradable organic carbon content, fraction of DOC decomposed, $CH_4$ fraction in landfill gas, and oxidation factor for landfilling; $C_{NG}$ and $C_{grid}$ are the natural gas (MJ/t dry sludge) and electricity (kWh/t dry sludge) consumption intensities of fuel production; and $EF_{NG}$ and $EF_{grid}$ are the emission factors of natural gas combustion and grid electricity. The factors $10^{-3}$, $10^{-6}$, and 16/12 convert g to kg, MJ to TJ, and C to $CH_4$, respectively. The incineration and landfilling factors are expressed per tonne of wet sludge, and the composting factors per tonne of dry solids.

#### 2.3.4. Indirect Emissions from Electricity Use (Scope 2)

The indirect emission from electricity use was calculated by multiplying the daily electricity consumption $E_{grid}$ (kWh/d) by the grid emission factor $EF_{grid}$ (Equation (13)) [36].

$$CO_{2,grid}\ (kg\ CO_2\text{-}eq/d) = E_{grid} \times EF_{grid} \quad (13)$$

#### 2.3.5. Credits from Resource Recovery

The credit for effluent reuse was calculated by applying the Environmental Product Declaration (EPD) emission factors of the substituted water to the reused volumes that replace tap water and industrial water (Equation (14)) [36]. The biogas credit was divided into grid electricity displaced by on-site generation and natural gas displaced by boiler use (Equations (15) and (16)); the calorific value of biogas and the generator and boiler efficiencies were taken from the literature [36–39]. The sludge recovery credits were calculated for the substitution of chemical fertilizer by compost [34] and of bituminous coal by solid fuel [35,40–42] with Equations (17) and (18). We calculated the coal substitution on an energy basis. The heat of the solid fuel was assumed to replace the same amount of heat from bituminous coal. The $CO_2$ from the combustion of the solid fuel was excluded as biogenic, as for incineration, and the $CH_4$ and $N_2O$ from its combustion were not included. The credits are attributional estimates calculated with the average emission factors

of the replaced products (Table 2). The net emissions are therefore conditional on these assumptions and are not measured reductions.

$$\text{Credit}_{\text{water}}\ (\text{kg CO}_2\text{-eq/d}) = Q_{\text{tapwater}} \times EF_{\text{tapwater}} + Q_{\text{indwater}} \times EF_{\text{indwater}} \quad (14)$$

$$\text{Credit}_{\text{grid}}\ (\text{kg CO}_2\text{-eq/d}) = V_{\text{grid}} \times f_{CH_4} \times \rho_{CH_4} \times NCV \times \eta_{\text{grid}} \times 1/3.6 \times EF_{\text{grid}} \quad (15)$$

$$\text{Credit}_{\text{NG}}\ (\text{kg CO}_2\text{-eq/d}) = V_{\text{boiler}} \times f_{CH_4} \times \rho_{CH_4} \times NCV \times \eta_{\text{boiler}} \times EF_{\text{NG}} \times 10^{-6} \quad (16)$$

$$\text{Credit}_{\text{com}}\ (\text{kg CO}_2\text{-eq/d}) = M_{\text{com}} \times (1 - MC/100) \times EF_{\text{com}} \quad (17)$$

$$\text{Credit}_{\text{SRF}}\ (\text{kg CO}_2\text{-eq/d}) = M_{\text{SRF}} \times (1 - MC/100) \times LHV_{\text{SRF}} \times EF_{\text{coal}} \times 10^{-3} \quad (18)$$

where $Q_{\text{tapwater}}$ and $Q_{\text{indwater}}$ are the daily reused volumes replacing tap water and industrial water (t/d), and $EF_{\text{tapwater}}$ and $EF_{\text{indwater}}$ are the emission factors per tonne of each water produced. $V_{\text{grid}}$ and $V_{\text{boiler}}$ are the daily biogas volumes used for on-site generation and in boilers ($m^3$/d); NCV is the net calorific value of $CH_4$; $\eta_{\text{grid}}$ and $\eta_{\text{boiler}}$ are the generator and boiler efficiencies; and 1/3.6 converts MJ to kWh. $EF_{\text{com}}$ is the avoided emission per tonne of dry sludge from the substitution of chemical fertilizer by compost, $LHV_{\text{SRF}}$ is the lower heating value of the solid fuel, and $EF_{\text{coal}}$ is the emission factor of bituminous coal combustion.

**Table 2.** Emission factors and parameters applied in the calculations.

| Symbol | Description | Value | IPCC Uncertainty Range | Source |
|---|---|---|---|---|
| $GWP_{100}$ (AR6) | $CO_2/CH_4$ (non-fossil)/$N_2O$ | 1/27/273 | – | [26] |
| $B_0$ | Maximum $CH_4$ producing capacity | 0.6 kg $CH_4$/kg BOD | – | [27] |
| $MCF_j$ | Methane correction factor, centralized aerobic treatment | 0.03 | 0.003–0.09 | [27] |
| $EF_{\text{treat}}$ | $N_2O$ emission factor, biological treatment | 0.016 kg $N_2O$-N/kg influent N | 0.00016–0.045 | [27] |
| $EF_{\text{effluent}}$ | $N_2O$ emission factor, effluent discharge | 0.005 kg $N_2O$-N/kg discharged N | 0.0005–0.075 | [27] |
| $f_{CH_4}$, $\rho_{CH_4}$ | $CH_4$ volume fraction and density of biogas (0 °C, 1 atm) | 0.6; 0.7168 kg/$m^3$ | – | [29] |
| $EF_{CH_4}$, $EF_{N_2O}$ (inc) | Sludge incineration (wet basis) | 9.7 g $CH_4$/t wet sludge; 900 g $N_2O$/t wet sludge | – | [30] |
| MCF, DOC, $DOC_f$, F, OX | Sludge landfilling (mass-balance method) | 0.6; 50 kg C/t wet sludge; 0.77; 0.5; 0 | MCF 0.4–1.0 | [31,32] |
| $EF_{CH_4,com}$, $EF_{N_2O,com}$ | Sludge composting (dry basis) | 10 kg $CH_4$/t dry solids; 0.6 kg $N_2O$/t dry solids | 0.08–20; 0.2–1.6 | [33] |
| $C_{NG}$, $C_{grid}$ | Natural gas and electricity use for SRF production (dry basis) | 1180 MJ/t dry solids; 77.78 kWh/t dry solids | – | [34] |
| $EF_{NG}$, $EF_{coal}$ | $CO_2$ emission factors of natural gas and bituminous coal combustion | 56,100; 94,600 kg $CO_2$/TJ | – | [35] |
| $EF_{grid}$ | Grid electricity emission factor (Korean EPD) | 0.495 kg $CO_2$-eq/kWh | – | [36] |
| $EF_{tapwater}$, $EF_{indwater}$ | Emission factors of tap water and industrial water production (EPD) | 0.237; 0.211 kg $CO_2$-eq/t water | – | [36] |
| NCV, $\eta_{grid}$, $\eta_{boiler}$ | Net calorific value of $CH_4$; efficiencies of biogas power generation and boiler | 48 MJ/kg $CH_4$; 0.35; 0.82 | – | [37–39] |
| $EF_{com}$ | Fertilizer substitution credit of compost (dry sludge basis) | 288 kg $CO_2$-eq/t dry solids | – | [34] |
| $LHV_{SRF}$ | Lower heating value of dried sludge fuel | 14.5 MJ/kg dry solids | – | [42] |

### *2.4. Statistical and Sensitivity Analyses*

Differences in emission intensity among process groups and capacity classes were tested with the Kruskal–Wallis test, because the plant-level distributions were skewed and the group sizes differed widely. We restricted the process-group test to the ten groups with at least ten plants (573 plants), at a significance level of 0.05. The annual totals include all 610 plants. Group values are reported as arithmetic means in the text, and the number of plants, medians, interquartile ranges, and test results are given in Supplementary Table S3. Dunn–Bonferroni comparisons and rank effect sizes are reported in Table S9. We also fitted a regression of the log intensity on capacity, utilization, influent BOD and T-N, the inflow-to-effluent ratio and process group, with HC3 confidence intervals. Two further models added effluent quality or kept all 610 plants. The coefficients are adjusted associations, not causal process effects.

The uncertainty arising from the choice of emission factors was examined with a one-at-a-time (OAT) sensitivity analysis. Seven parameters were selected because they drive the main sources or have wide IPCC uncertainty ranges: (1) the $N_2O$ emission factor for biological treatment, (2) the $N_2O$ emission factor for effluent discharge, (3) the MCF for aerobic treatment, (4) the MCF for landfilling, (5) the grid emission factor, (6) the vented fraction of surplus biogas, and (7) the $CH_4$ fraction of biogas. The ranges followed the IPCC uncertainty ranges where available and the assumed ranges listed in Supplementary Table S4 otherwise [27,31]. Because each parameter enters the equations linearly, the affected emission term of each plant was rescaled by the parameter ratio, and the resulting change in the mean net emission intensity of the 610 plants was compared. Apart from the landfill MCF, the parameters of sludge treatment and of the resource recovery credits were not varied, because these items account for only 4.8% and 4.6% of the mean gross emission intensity, which limits their possible influence. The same analysis was applied to the annual gross emissions of the 610 plants.

### *2.5. Use of Generative AI Tools*

During the preparation of this manuscript, the authors used Claude (Anthropic, San Francisco, CA, USA; Fable 5.1 and Fable 5.2; accessed October and November 2026) for the following purposes: (i) translation of the Korean manuscript into English and subsequent language editing; (ii) formatting of the manuscript, tables, and reference list to the journal template; and (iii) cross-checking the consistency of numerical values between the manuscript text, tables, figures, and the input dataset extracted from the national sewerage statistics. In the revision, the tool was also used for (iv) drafting and language editing of the revised text and of the responses to the reviewers, and (v) recalculating the results after the corrections to the calculation, running the statistical tests, and redrawing the figures and the Supplementary tables with scripts; the authors checked these outputs against the input dataset. The tool was not used for study design, data extraction, or selection of emission factors. The core prompts used were as follows: "Translate the following Korean manuscript into academic English suitable for the journal Water, following the sentence structure and terminology of the attached reference papers by the author, and avoiding generic AI-style phrasing"; "Restructure the manuscript into the Water format (Abstract, Introduction, Materials and Methods, Results and Discussion, Conclusions) and convert the reference list to the MDPI citation style"; and "Compare all numerical values in the English manuscript with those in the Korean manuscript and the input dataset, and report any discrepancies." For the revision, the core prompt was: "Apply the corrections to the plant-level results, regenerate the figures and the Supplementary tables, and report any value that differs from the manuscript." All outputs were reviewed, verified, and edited by the authors.

## 3. Results and Discussion

### *3.1. Operating Characteristics of Public WWTPs in Korea*

The treatment processes, plant capacities, and installed facilities of the 610 plants were surveyed to characterize their operation. These characteristics serve as the reference for comparing the operational emissions in the following sections.

Figure 3a shows the distribution of treatment processes. Among the 610 plants, BNR was the most common process (165 plants, 27.0%), followed by SBR (159 plants, 26.1%). The large share of BNR indicates that advanced processes capable of removing nitrogen and phosphorus, introduced in response to stricter effluent standards, have become the mainstream process of public WWTPs in Korea. MBR, AS, biofilm, and various combined processes were also in use, so a wide range of biological processes is operated nationwide.

Differences in reactor configuration and operating mode among these processes affect the operational GHG emissions. BNR requires internal recirculation and additional aeration for nitrification and denitrification; aeration is the largest electricity consumer at a WWTP [3], and $N_2O$ can be produced as a by-product of nitrification and denitrification [11,43]. MBR also requires continuous aeration to control membrane fouling, and aeration energy is known to account for most of its operating energy [44]. SBR runs the treatment steps sequentially in a single reactor; $N_2O$ can be released while the reactor alternates between aerobic and anoxic conditions for nitrogen removal [45], and a study of a full-scale SBR reported that $N_2O$ accounted for more than 60% of the plant's carbon footprint [46]. Biofilm processes, which rely on attached microorganisms, produce relatively little sludge and remove organic matter stably [47], and some configurations have been reported to reduce $N_2O$ emissions by about 44% through microbial activity [48]. The treatment process is thus a major determinant of both energy consumption and direct GHG emissions, and these process characteristics were used to interpret the differences in operational emissions.

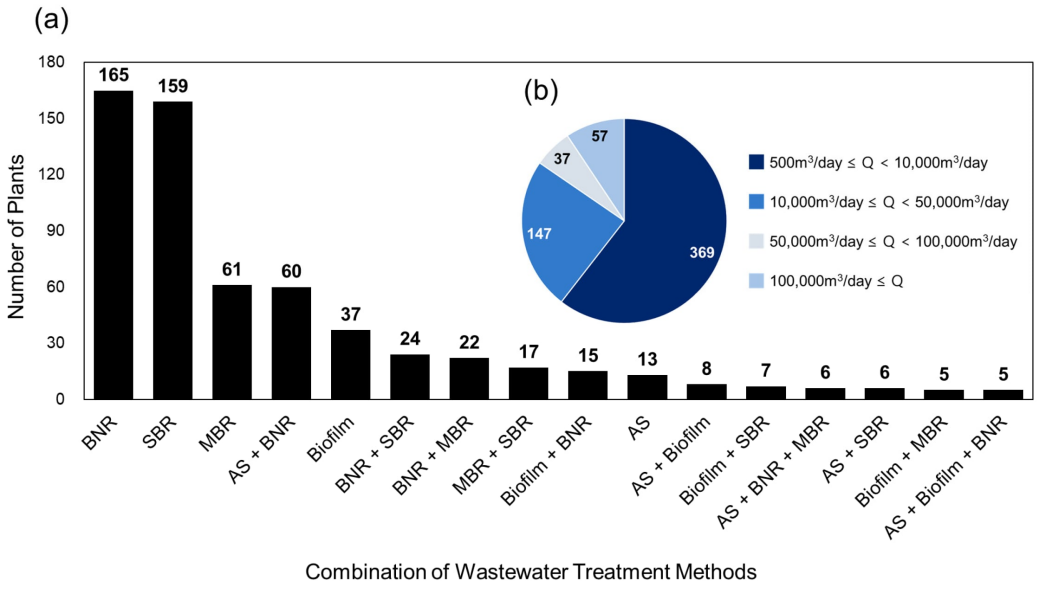


**Figure 3.** Characteristics of the investigated wastewater treatment plants: (**a**) distribution of treatment processes; (**b**) distribution of plants by treatment capacity. AS, activated sludge; BNR, biological nutrient removal; MBR, membrane bioreactor; SBR, sequencing batch reactor.

Figure 3b shows the distribution of plant capacity. Plants with a capacity of 500 to <10,000 $m^3$/d were the most numerous (369 plants, 60.5%), whereas large plants of ≥100,000 $m^3$/d numbered 57 (9.3%). Small and medium-sized plants therefore make up most of the public WWTPs in Korea by number.

Table 3 lists the plants with advanced treatment, sludge treatment, water reuse, anaerobic digestion, and sludge resource recovery. The composition of these facilities affects not only the magnitude of the operational emissions and the contribution of each source but also the environmental credits from resource recovery. About 71% of the plants operate advanced treatment, so nitrogen removal takes place at most of them. Direct $N_2O$ emissions from nitrification and denitrification (Scope 1) and the electricity used for aeration and internal recirculation (Scope 2) are therefore likely to be the main sources of operational emissions. Water reuse was reported by 93.6% of the plants (571), but only 19.0% (116) had a dedicated reuse facility, indicating that most reuse takes place within the plant as on-site process water. Anaerobic digestion, by contrast, was installed at only 10.2% of the plants (62), so energy recovery from biogas is confined to a small number of plants; of these, 58 reported biogas production and 44 reported surplus biogas. Sludge was sent to composting or fuel production, including outsourced treatment, by 54.1% of the plants (330), so the fertilizer and fossil fuel substitution achieved through these routes is expected to vary widely among plants. The main GHG sources are thus common to most plants, whereas the type and size of the environmental credits depend on which resource recovery facilities a plant has. This difference in facility composition is an important background for interpreting the gross and net emission intensities of each process group, because plants with the same treatment process can still differ in net emissions according to their resource recovery facilities.

**Table 3.** Number and percentage of the 610 public wastewater treatment plants with advanced treatment, sludge treatment, water reuse, anaerobic digestion, and sludge resource recovery.

| Facility or Activity | Plants (n) | Share of 610 Plants (%) |
|---|---|---|
| Advanced treatment (nitrogen and phosphorus removal) | 432 | 70.8 |
| Sludge treatment facility | 486 | 79.7 |
| Water reuse reported | 571 | 93.6 |
| with a dedicated reuse facility | 116 | 19.0 |
| Anaerobic digestion | 62 | 10.2 |
| with biogas production reported | 58 | 9.5 |
| with surplus biogas reported | 44 | 7.2 |
| Sludge resource recovery (composting or SRF, including outsourced treatment) | 330 | 54.1 |

### *3.2. Operational Emissions by Treatment Process*

The operational emissions were compared among treatment processes to identify the emission characteristics and main sources of each (Figure 4). Scope 1 was divided into biological wastewater treatment (WT), sludge treatment (ST), and the direct gas emission of surplus digester gas (DGE; the value calculated under the assumption of complete venting described in Section 2.3.2), and the contribution of each source was used to explain the differences among processes. The comparison was confined to Scope 1 and Scope 2; the credits from anaerobic digestion, reuse, and sludge recovery were excluded. Each group value is the arithmetic mean of the plant-level values per functional unit, and the number of plants, the number with digesters, and the medians and interquartile ranges of each group are given in Supplementary Table S3.

Figure 4 shows the operational emission intensity of each process divided into these sources. A process group marked as including anaerobic digestion means that some of the plants in that group have digesters, not that all of them do. The mean gross emission intensity of the 610 plants was 1.039 kg $CO_2$-eq/$m^3$, of which Scope 2 accounted for 53.2%, direct emissions from wastewater treatment for 40.8% ($N_2O$ 31.6%, $CH_4$ 9.3%), sludge treatment for 4.8%, and DGE for 1.1%. The gross emission intensity by process ranged from 0.696 to 1.495 kg $CO_2$-eq/$m^3$, a difference of about 2.1-fold between the highest and

lowest groups. Among the single processes, MBR had the highest emission intensity, and the biofilm–MBR and BNR–MBR combinations were also relatively high at 1.343 and 1.196 kg $CO_2$-eq/$m^3$, respectively. The AS–biofilm–BNR combination was the lowest of all groups at 0.696 kg $CO_2$-eq/$m^3$, about 47% of the MBR value. Scope 2 varied about 3.6-fold among processes (0.264–0.955 kg $CO_2$-eq/$m^3$), whereas Scope 1 varied comparatively little (0.409–0.578 kg $CO_2$-eq/$m^3$). This indicates that the energy consumption characteristics of each process are the main factor behind the differences in operational emissions.

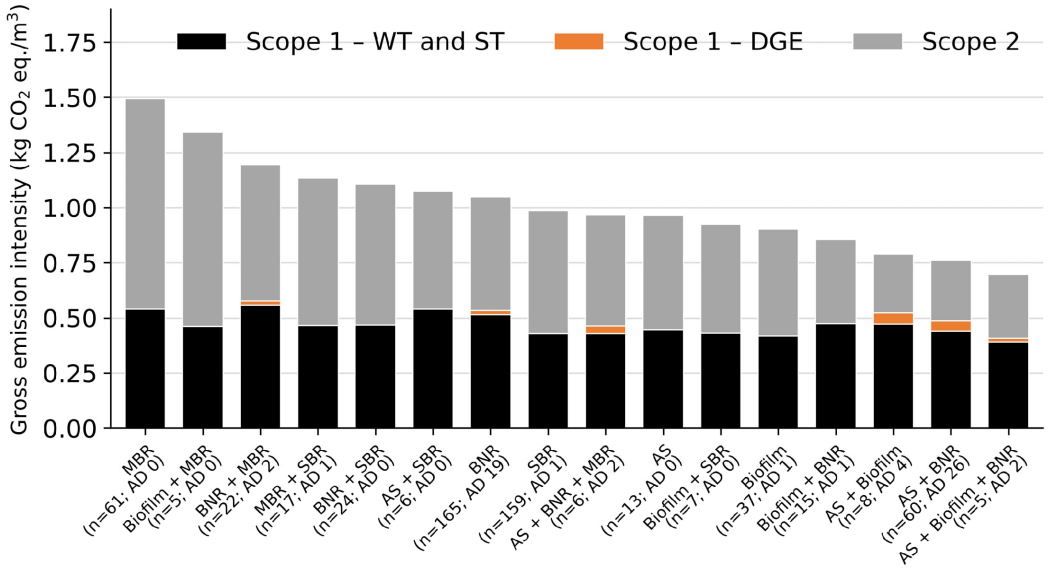


**Figure 4.** Operational emission intensity by treatment process based on Scope 1 and Scope 2 emissions. Values are arithmetic means of the plant-level values in each group; n, number of plants; AD, number of plants with reported digester operation (59 of the 62 plants with digesters; 58 of them reported biogas production); the underlying values are listed in Supplementary Table S5. Medians and interquartile ranges are given in Supplementary Table S3. WT and ST, water-treatment- and sludge-treatment-based emissions; DGE, direct gas emission of surplus biogas from anaerobic digestion, assumed to be vented; AS, activated sludge; BNR, biological nutrient removal; MBR, membrane bioreactor; SBR, sequencing batch reactor.

For MBR, Scope 1 and Scope 2 were 0.541 and 0.955 kg $CO_2$-eq/$m^3$, respectively; its Scope 2 was the highest of all processes and accounted for 63.8% of its gross emission intensity, making MBR the single process with the highest operational emissions. MBR is known to be energy-intensive because continuous aeration is needed to control membrane fouling and sustain the biological process [44], and the present results likewise show a higher Scope 2 than the other single processes. BNR and SBR were close to the overall mean. The AS–BNR, AS–biofilm, and AS–biofilm–BNR combinations had low Scope 2 values (0.264–0.287 kg $CO_2$-eq/$m^3$) and belonged to the group with the lowest gross emission intensity; the AS–biofilm–BNR combination had the lowest Scope 1 of all groups (0.409 kg $CO_2$-eq/$m^3$), and the AS–biofilm combination had the lowest Scope 2 (0.264 kg $CO_2$-eq/$m^3$). The Kruskal–Wallis test for the ten groups with at least ten plants showed significant differences in gross emission intensity and Scope 2 among process groups ($p < 0.001$), but not in sludge treatment emissions ($p = 0.47$; Table S9). After adjustment for capacity, utilization rate and influent strength, the intensity of MBR remained higher than that of BNR (ratio 1.228, 95% CI 1.145–1.317; Table S9). This is an association and not a causal process effect.

The $CH_4$ emission from surplus digester gas (DGE) was calculated for the 44 plants that reported surplus biogas among the 62 plants with digesters. Averaged over each process group, it amounted to 0.001–0.053 kg $CO_2$-eq/$m^3$, or 0.1–6.7% of the operational

emissions (Figure 5). For the 44 plants themselves, however, the mean DGE was 0.165 kg $CO_2$-eq/$m^3$ (maximum 0.746), equivalent to 16.5% of their gross emission intensity on average (maximum 57.0%). Surplus biogas released without treatment therefore has a small effect on the group means but can be a substantial part of the operational emissions of an individual plant with a digester. This value was obtained by assuming, in the absence of plant-level flare records, that all unused surplus biogas is vented directly; whether and how much is actually released cannot be confirmed from the Sewerage Statistics. If surplus biogas is combusted, most of the $CH_4$ is oxidized, and the $CH_4$ actually emitted and its contribution to the emissions would be lower than estimated here. Field measurements of flares at WWTPs have reported $CH_4$ destruction efficiencies of 95.1–99.8% [49]. The emission estimated here should therefore be read as a conservative value that does not account for flaring, and incorporating plant-level data on the direct release and flaring of surplus biogas would improve the accuracy of the estimate.

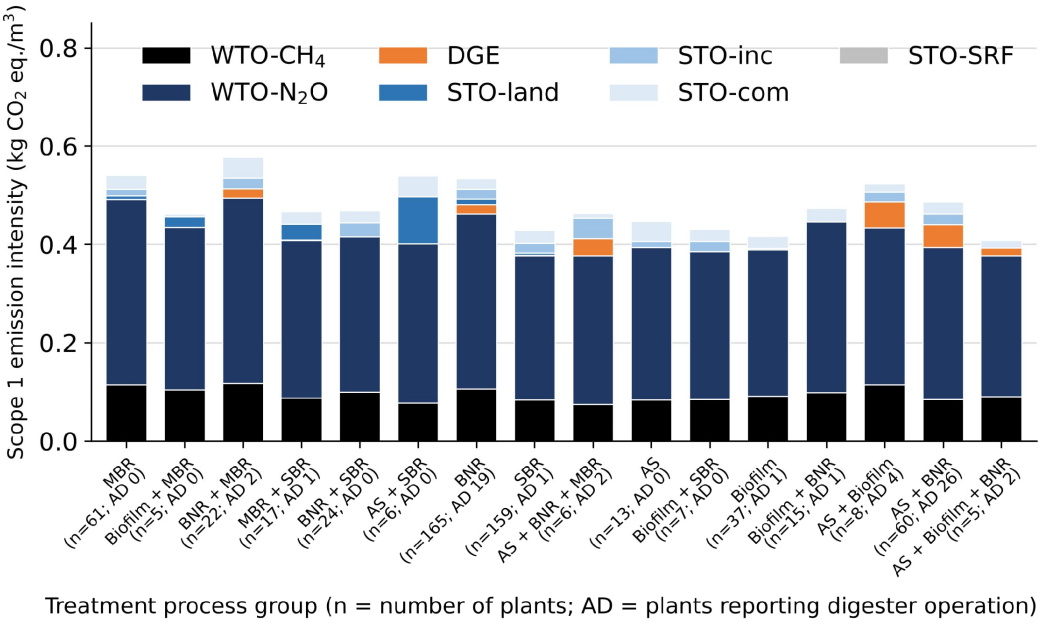


**Figure 5.** Breakdown of Scope 1 emission intensity by treatment process according to wastewater treatment and sludge management sources. Values are arithmetic means of the plant-level values in each group and are listed in Supplementary Table S5. Medians and interquartile ranges are given in Supplementary Table S3. WTO-$CH_4$ and WTO-$N_2O$, wastewater-treatment-originated emissions from $CH_4$ and $N_2O$; DGE, direct gas emission of surplus biogas from anaerobic digestion, assumed to be vented; STO-land, STO-inc, STO-com, and STO-SRF, sludge-treatment-originated emissions from landfilling, incineration, composting, and solid refuse fuel production; AS, activated sludge; BNR, biological nutrient removal; MBR, membrane bioreactor; SBR, sequencing batch reactor.

Figure 5 breaks the Scope 1 emission intensity of each process down into wastewater treatment (WTO), the direct $CH_4$ emission from digesters (DGE), and sludge treatment (STO). $N_2O$ from wastewater treatment was the largest single source in most processes. The emission intensity from $N_2O$ was 0.286–0.378 kg $CO_2$-eq/$m^3$, about 2.8–4.2 times the emission intensity from $CH_4$ (0.075–0.117 kg $CO_2$-eq/$m^3$). The $N_2O$ emission intensity of MBR and of the BNR–MBR combination, at 0.378 and 0.377 kg $CO_2$-eq/$m^3$, was higher than that of the other processes. Because the same emission factors were applied to every plant, the differences in $N_2O$ and $CH_4$ emission intensities among groups reflect differences in the influent T-N and BOD loads per unit of treated water, and these two groups received relatively high T-N loads. The sludge treatment emissions ranged from 0.016 (AS–biofilm–BNR) to 0.138 kg $CO_2$-eq/$m^3$ (AS–SBR), or about 4–26% of Scope 1.

The sludge treatment emissions differed among processes, but this difference cannot be attributed to the wastewater treatment process itself. Landfilling, incineration, composting, and fuel production are final disposal routes chosen by each plant rather than steps determined by the treatment process, and the sludge treatment emissions depend on the sludge quantity and moisture content as well as on the accounting factors of each route.

To separate these effects, Table 4 lists, for each of the four disposal routes, the number of plants using it, the mean sludge treatment emissions of those plants, and the emission and recovery credit factors per tonne of dry solids. The emission factors were 2230 kg $CO_2$-eq/t dry solids for landfilling, 1319 for incineration, 434 for composting, and 105 for SRF production. The landfilling and incineration factors are given per tonne of wet sludge, and we converted them with the median moisture content (81.35%). Landfilling was used by only 22 of the 610 plants and received only 0.7% of the sludge sent to the four routes, but because the $CH_4$ generated by anaerobic decomposition is assumed to be released without recovery, it had the largest factor per unit mass, and the plants using it also had the highest mean sludge treatment emissions (0.195 kg $CO_2$-eq/$m^3$). Incineration was used by 80 plants; once biogenic $CO_2$ was excluded, $N_2O$ from combustion made up most of its emission. Composting, used by 310 plants, was the most common route, with a direct emission factor about 33% of that of incineration. A route with a high accounting factor can therefore contribute substantially to the gross emission intensity even when the sludge quantity is small.

**Table 4.** Comparison of sludge disposal pathways: plants using each pathway, their mean modeled route intensity, and direct-emission, credit, and net factors per tonne of dry solids.

| Pathway | Plants Using | Mean Modeled Route Intensity of Users (kg $CO_2$-eq/$m^3$) | Emission (kg $CO_2$-eq/t Dry Solids) | Credit (kg $CO_2$-eq/t Dry Solids) | Net (kg $CO_2$-eq/t Dry Solids) |
|---|---|---|---|---|---|
| Landfilling [a] | 22 | 0.195 | 2229.5 | 0.0 | +2229.5 |
| Incineration [a] | 80 | 0.130 | 1318.8 | 0.0 | +1318.8 |
| Composting [b] | 310 | 0.050 | 433.8 | 288.0 | +145.8 |
| SRF [c] | 65 | 0.005 | 104.7 | 1371.7 | −1267.0 |

Note(s): [a] Converted from the wet-basis IPCC factors with a moisture content of 81.35%. [b] IPCC dry-basis factors (10 kg $CH_4$ and 0.6 kg $N_2O$ per tonne of dry solids); the credit is the substitution of chemical fertilizer. [c] Natural gas (1180 MJ/t dry solids) and electricity (77.78 kWh/t dry solids) used for SRF production; the credit is the substitution of bituminous coal (14.5 MJ/kg dry, 94,600 kg $CO_2$/TJ). Auxiliary fuel and electricity are not included for landfilling, incineration and composting.

The AS–SBR combination, which had the highest sludge treatment emissions (0.138 kg $CO_2$-eq/$m^3$), was strongly affected by two of its six plants using landfilling, whereas the AS–biofilm–BNR combination with the lowest value (0.016 kg $CO_2$-eq/$m^3$) used almost no route other than composting. The differences in sludge treatment emissions in Figure 5 thus reflect plant-specific sludge generation, moisture content, and disposal routes rather than the wastewater treatment process itself. Therefore, sludge-related emissions should be interpreted separately from the intrinsic emission characteristics of each process when operational emissions are compared among processes.

The sludge treatment emissions depended strongly on the sludge quantity per unit of treated water and on the accounting factor of the final disposal route. Shifting sludge from landfilling and incineration to recovery routes is therefore a possible way to reduce these emissions. We did not evaluate its effect here.

### *3.3. Change in Net Emissions with Resource Recovery (Biogas, Reused Water, and Sludge Recovery)*

The net emission intensity of each process was compared after incorporating the environmental credits from biogas use, effluent reuse, and sludge recovery, and the contribution

of each credit item to the reduction was analyzed (Figure 6). The difference between the gross and net emission intensities was used to evaluate how much resource recovery reduces operational emissions.

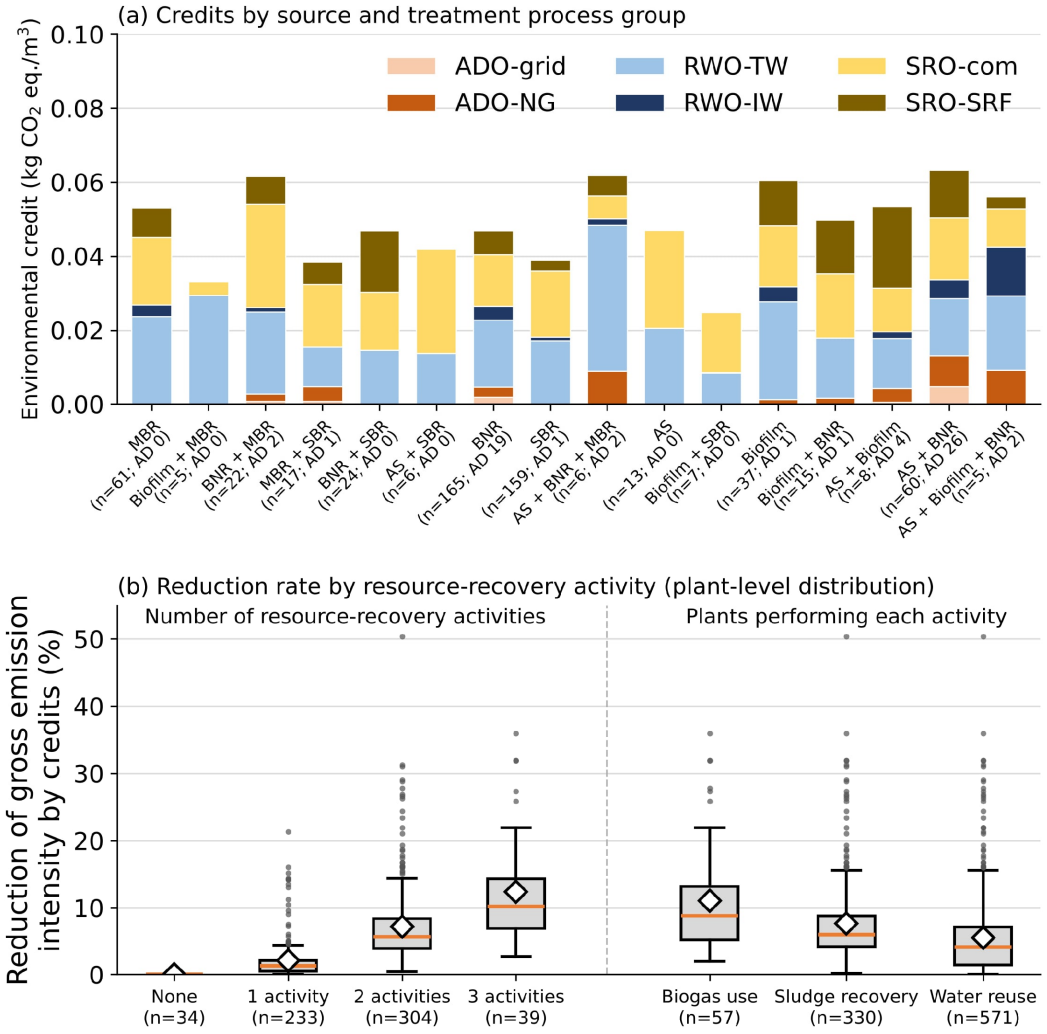


**Figure 6.** Environmental credits from resource recovery: (**a**) mean credit by source and treatment process group (arithmetic means of the plant-level values; the corresponding net emission intensity values and reduction rates are listed in Supplementary Table S6); (**b**) distribution of the plant-level reduction of gross emission intensity by credits according to the number of resource recovery activities performed (biogas use, sludge recovery, water reuse) and for the plants performing each activity (box, interquartile range; line, median; diamond, mean; whiskers, 1.5 × IQR; dots, outliers). ADO-grid and ADO-NG, anaerobic-digestion-originated credits for grid electricity and natural gas; RWO-TW and RWO-IW, reclaimed-water-originated credits for tap water and industrial water; SRO-com and SRO-SRF, sludge-recovery-originated credits for composting and solid refuse fuel; AS, activated sludge; BNR, biological nutrient removal; MBR, membrane bioreactor; SBR, sequencing batch reactor.

Figure 6a shows the environmental credits of each process by source. The mean credit of the 610 plants was 0.048 kg $CO_2$-eq/$m^3$, equivalent to 4.6% of the gross emission intensity; this value is the mean credit divided by the mean gross emission intensity, whereas the reduction rates reported below for each resource recovery activity are arithmetic means of the plant-level reductions (credit/gross emission intensity). The mean credit by process group was 0.025–0.063 kg $CO_2$-eq/$m^3$, corresponding to a 2.5–8.3% reduction of the gross emission intensity. Biogas credits appeared only in groups with a high proportion of digesters, such as AS–BNR and AS–biofilm–BNR; in most groups, including SBR and BNR,

reuse and composting credits made up most of the total. MBR remained the highest after credits, with a net emission intensity of 1.442 kg $CO_2$-eq/$m^3$, and the AS–biofilm–BNR combination the lowest at 0.640 kg $CO_2$-eq/$m^3$ (Supplementary Table S6). The size of the credit was determined less by the treatment process than by whether a plant has, and actually uses, resource recovery facilities (Figure 6b). The 576 plants performing at least one of biogas use, water-substituting reuse, and sludge recovery had a mean reduction of 5.5%, whereas the 34 plants performing none had 0%, and the reduction rose with the number of activities, from 2.1% for one activity (233 plants) to 7.2% for two (304 plants) and 12.3% for three (39 plants). By activity, the 57 plants using biogas had the highest reduction (11.1%), followed by the 330 plants recovering sludge (7.7%) and the 571 plants with water-substituting reuse (5.5%).

By item, the plant-mean credit consisted of sludge recovery (49.7%; fertilizer substitution by compost 34.9%, coal substitution by solid fuel 14.8%), effluent reuse (43.8%; tap water 38.6%, industrial water 5.2%), and biogas use (6.4%; natural gas 4.3%, electricity 2.1%). The mean credit of the plants performing each activity was 0.023 kg $CO_2$-eq/$m^3$ for effluent reuse (571 plants), 0.044 for sludge recovery (330 plants), and 0.033 for biogas use (57 plants). Reuse and sludge recovery dominate the plant mean because many plants perform them. Biogas use is limited to a small number of plants. On the annual basis, the shares of sludge recovery, effluent reuse and biogas use were 38.2%, 34.1% and 27.7%, respectively, because biogas is used mainly at large plants. Comparing the sludge recovery credits with the emission factors in Table 4, the fertilizer substitution credit of composting (288 kg $CO_2$-eq/t dry solids) offsets about 66% of its emission factor (434 kg $CO_2$-eq/t dry solids). For SRF production, the coal substitution credit (1372 kg $CO_2$-eq/t dry solids) was much larger than the emission factor (105 kg $CO_2$-eq/t dry solids). On the basis of these factors, the net emission decreases in the order of landfilling > incineration > composting > SRF production. The routes, however, do not include the same energy inputs (Section 2.3.3). Sludge management should therefore be evaluated on both its direct emissions and its substitution effects, and expanding sludge recovery is a candidate strategy for lowering the net operational emissions.

The water substitution credit from effluent reuse was the largest single credit item, but it was small relative to how widely reuse was reported. Reuse was reported by 93.6% of the 610 plants (Table 3), yet the reused volume credited with water substitution amounted to only about 9.4% of the total effluent. The rest of the reused volume is used for recreational water and stream flow maintenance, which were not counted as substitution. The size of the reuse credit is thus governed by the reuse rate that actually substitutes other water sources and by the mix of end uses rather than by the presence of a reuse facility, and expanding substitution uses such as industrial water supply appears to be the practical way to increase it.

At the 58 plants reporting biogas production, the direct $CH_4$ emission from surplus biogas (DGE) averaged 0.125 kg $CO_2$-eq/$m^3$, whereas the electricity and natural gas substitution credit from biogas use averaged 0.033 kg $CO_2$-eq/$m^3$; under the assumption that all surplus biogas is vented, the direct emission from digesters was about 3.8 times the biogas credit. The total credit of these plants, including reuse and sludge recovery, averaged 0.080 kg $CO_2$-eq/$m^3$, or 10.2% of their gross emission intensity (0.784 kg $CO_2$-eq/$m^3$), which is larger than the overall mean of 4.6%. Anaerobic digestion should therefore be evaluated on its direct emissions and its energy recovery credits together, and without flaring or greater use of surplus biogas, installing a digester alone cannot be expected to reduce net emissions. This comparison holds only if the surplus biogas is vented. It reverses when about 74% or more of the $CH_4$ is destroyed. With the default efficiency of an open flare (50%) [50], the remaining $CH_4$ (0.062 kg $CO_2$-eq/$m^3$) still exceeds the biogas credit. With

that of an enclosed flare (90%), it falls to 0.012 kg $CO_2$-eq/$m^3$, below the credit (Table S7). These are scenarios, because flare operation is not recorded in the statistics.

Venting 1 $m^3$ of biogas emits 11.6 kg $CO_2$-eq as $CH_4$, whereas using the same volume for on-site generation yields an electricity substitution credit of 0.99 kg $CO_2$-eq and using it in a boiler yields a natural gas substitution credit of 0.95 kg $CO_2$-eq. If all surplus biogas were used for power generation, the mean DGE of 0.125 kg $CO_2$-eq/$m^3$ would be eliminated and about 0.011 kg $CO_2$-eq/$m^3$ of credit would be added, lowering the net emission intensity of these plants by about 0.136 kg $CO_2$-eq/$m^3$. Because this benefit comes from avoiding direct $CH_4$ release rather than from energy recovery, the use of surplus biogas matters more as a methane control measure than as a means of energy self-sufficiency.

Anaerobic digestion stabilizes sludge and reduces its organic content while recovering biogas as an energy source. Previous studies have likewise reported that anaerobic digestion lowers the organic content of sludge, that the biogas produced can be used for electricity and heat [51], and that energy recovery from biogas reduces the external energy dependence of WWTPs, raises their energy self-sufficiency, and lowers operating costs [52]. Therefore, a full assessment of anaerobic digestion should weigh sludge reduction, energy production, and economic benefits alongside the direct $CH_4$ emissions and the emission offset from energy recovery. El Abbadi et al., who built an emission inventory of 15,863 WWTPs in the United States, also reported that methane leaking from anaerobic digesters outweighs the benefit of on-site power generation [19]. Their estimate concerns fugitive methane, not the surplus venting assumed here, so it does not validate the present estimate.

### *3.4. National Total Emissions and Emission Intensity by Plant Capacity*

Table 5 gives the annual emissions of the 610 plants and their emission intensity calculated in two ways. The plant mean is the average of the 610 plant-level intensities, so every plant counts equally regardless of its size. The flow-weighted value is the total annual emission divided by the total annual effluent volume, so a plant counts in proportion to the volume it treats. The annual gross emissions were 5.207 Mt $CO_2$-eq, and the flow-weighted intensity was 0.723 kg $CO_2$-eq/$m^3$, about 30% lower than the plant mean of 1.039 kg $CO_2$-eq/$m^3$. The flow-weighted intensity and the annual total describe the emissions of the sector as a whole. The plant mean describes a typical plant. The 610 plants treat 96.2% of the national effluent, so the annual total approximates the operational emissions of the public wastewater sector. We take it as the primary result. Receiving-water $N_2O$ accounted for 0.147 Mt $CO_2$-eq (2.8%) of this total. Without it, the annual gross emissions were 5.060 Mt $CO_2$-eq and the flow-weighted intensity was 0.702 kg $CO_2$-eq/$m^3$.

The ranking of the emission sources depends on which of the two values is used. On the annual basis, $N_2O$ from treatment and receiving water was the largest source (43.5%), followed by electricity (31.8%), treatment $CH_4$ (11.9%), $CH_4$ from surplus biogas (7.3%) and sludge treatment (5.5%). On the plant-mean basis, electricity was the largest (53.2%) and $N_2O$ the second (31.6%). The gap of 0.316 kg $CO_2$-eq/$m^3$ between the plant mean and the flow-weighted value came almost entirely from electricity (0.553 versus 0.230 kg $CO_2$-eq/$m^3$). The on-site emissions were nearly the same (0.435 versus 0.453 kg $CO_2$-eq/$m^3$). The reason is the relation between electricity intensity and plant capacity shown in Figure 7.

Figure 7 shows the plant-level distributions of gross emission intensity and Scope 2 by the capacity classes defined in Section 2.2. The mean gross emission intensity decreased with plant capacity, from 1.137 kg $CO_2$-eq/$m^3$ in the 500–10,000 $m^3$/d class to 0.964 in the 10,000–50,000 class, 0.841 in the 50,000–100,000 class, and 0.720 kg $CO_2$-eq/$m^3$ in the ≥100,000 $m^3$/d class (Kruskal–Wallis test, $p < 0.001$; Supplementary Table S3b). Most of this decrease came from Scope 2: the electricity-related emission intensity fell about 3.2-fold, from 0.687 kg $CO_2$-eq/$m^3$ in the smallest class to 0.212 kg $CO_2$-eq/$m^3$ in the largest, and the

spread within each class narrowed as well (Figure 7b), whereas the direct emission from wastewater treatment showed no consistent trend with capacity (0.401–0.481 kg $CO_2$-eq/$m^3$; Supplementary Table S3b). No plant below 10,000 $m^3$/d had a digester, whereas 39 of the 57 plants of ≥100,000 $m^3$/d did, so DGE was concentrated in the largest class. These results also show that plant capacity is confounded with the process-group differences in Figure 4: 42 of the 61 MBR plants were below 10,000 $m^3$/d, whereas 27 of the 60 plants with the low-emission AS–BNR combination were large plants of ≥50,000 $m^3$/d.

**Table 5.** Annual emissions of the 610 plants and their emission intensity, expressed as the plant mean and as the flow-weighted value, by emission source and by emission location.

| Component | Annual Emissions (Mt $CO_2$-eq/year) | Share of Gross (%) | Plant Mean (kg $CO_2$-eq/$m^3$) | Flow-Weighted (kg $CO_2$-eq/$m^3$) |
|---|---|---|---|---|
| By emission source | | | | |
| $N_2O$ (treatment and receiving water) | 2.267 | 43.5 | 0.328 | 0.315 |
| $CH_4$ (treatment) | 0.621 | 11.9 | 0.096 | 0.086 |
| $CH_4$ (surplus biogas) [a] | 0.380 | 7.3 | 0.012 | 0.053 |
| Sludge treatment [b] | 0.284 | 5.5 | 0.050 | 0.039 |
| Electricity [c] | 1.655 | 31.8 | 0.553 | 0.230 |
| By emission location | | | | |
| On-site [d] | 3.261 | 62.6 | 0.435 | 0.453 |
| Plant electricity | 1.652 | 31.7 | 0.552 | 0.229 |
| Outsourced sludge treatment [e] | 0.147 | 2.8 | 0.037 | 0.020 |
| Receiving-water $N_2O$ | 0.147 | 2.8 | 0.014 | 0.020 |
| Gross emissions | 5.207 | 100 | 1.039 | 0.723 |
| Credits (conditional) | 0.457 | 8.8 | 0.048 | 0.063 |
| Net emissions | 4.750 | 91.2 | 0.991 | 0.659 |

Note(s): [a] Complete venting assumed. [b] Excluding electricity. [c] Plant electricity and electricity for SRF production. [d] Treatment $CH_4$ and $N_2O$, surplus biogas $CH_4$, and self-treated sludge. [e] Including fuel and electricity for SRF production. The flow-weighted intensity is the annual emission divided by the annual effluent volume ($7.204 \times 10^9$ $m^3$).

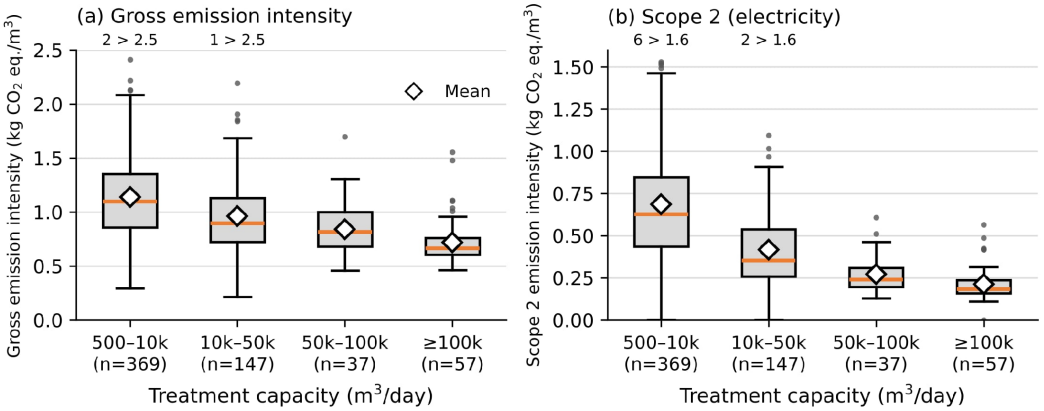


**Figure 7.** Distribution of plant-level (**a**) gross emission intensity and (**b**) Scope 2 emission intensity by treatment capacity class (box, interquartile range; line, median; diamond, mean; whiskers, 1.5 × IQR; dots, outliers; the number of plants above the axis limit is given at the top). Mean values of each emission source by class are listed in Supplementary Table S3b.

Plants below 10,000 $m^3$/d made up 60.5% of the plants but produced 5.9% of the annual gross emissions. Plants of ≥100,000 $m^3$/d made up 9.3% of the plants and produced 67.3%. The emission intensity is therefore highest at small plants, where electricity efficiency is the main issue. The national total, however, is decided by $N_2O$, electricity and surplus biogas at large plants. The official 2024 inventory reports 1.411 Mt $CO_2$-eq for domestic

wastewater [53]. This study gives 2.888 Mt $CO_2$-eq for treatment $CH_4$ and $N_2O$ and receiving-water $N_2O$. Most of the difference is in $N_2O$ (2.267 versus 0.975 Mt $CO_2$-eq), and $CH_4$ differs less (0.621 versus 0.436 Mt $CO_2$-eq; Table S10). The main reason is the method. The inventory follows the 2006 IPCC Guidelines, whereas this study uses the Tier 1 factors of the 2019 Refinement, in which the default $N_2O$ emission factor for treatment plants is much higher. The plant coverage and the GWP values also differ, so the two values cannot be compared directly. For comparison, Chen et al. reported a mean intensity of 0.45 kg $CO_2$-eq/$m^3$ for 247 WWTPs in China [20].

### *3.5. Sensitivity Analysis*

Figure 8 ranks the parameters selected in Section 2.4 by their effect on the mean net emission intensity of the 610 plants (base value 0.991 kg $CO_2$-eq/$m^3$).

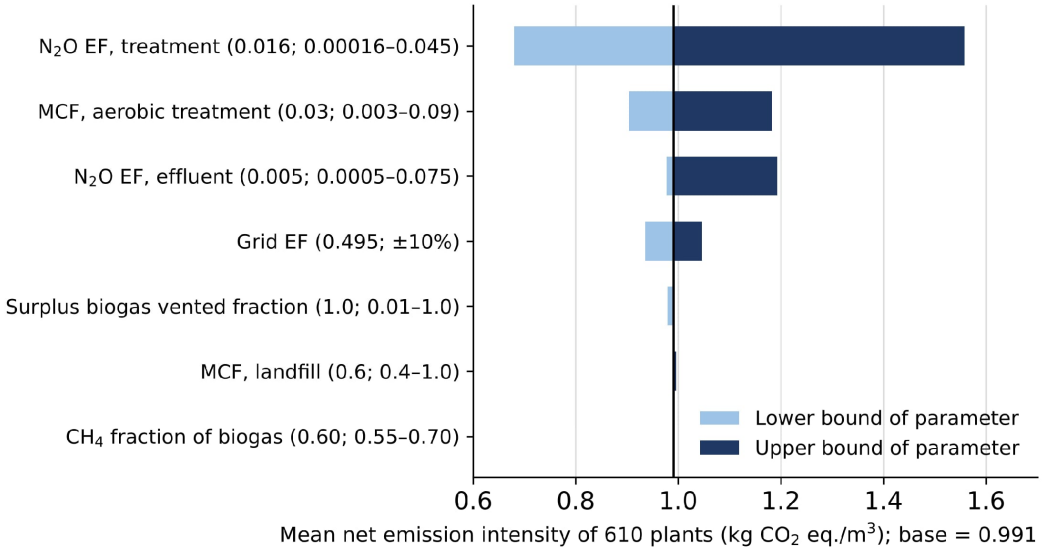


**Figure 8.** One-at-a-time sensitivity of the mean net emission intensity of the 610 plants to emission factors and assumptions over the parameter ranges given in Supplementary Table S4 (numbers in parentheses: base value; range). The resulting values are listed in Supplementary Table S4.

The most influential parameter was the $N_2O$ emission factor for biological treatment: between the lower and upper bounds of the IPCC range, the mean net emission intensity varied from 0.680 to 1.559 kg $CO_2$-eq/$m^3$ (−31% to +57%), followed by the MCF for aerobic treatment (−9% to +19%), the $N_2O$ emission factor for effluent discharge (−1.3% to +20.5%), and the grid emission factor (±5.6%). By contrast, assuming that 99% of surplus biogas is combusted lowered the mean net emission intensity by only 1.2%, and the landfill MCF and the $CH_4$ fraction of biogas changed it by 0.5% or less. The annual total is the primary result, so we repeated the analysis for the annual gross emissions (base value 5.207 Mt $CO_2$-eq; Table S4). The flow-weighted intensity changes by the same percentages. Over the IPCC range of the $N_2O$ emission factor for biological treatment, the annual gross emissions varied from 3.11 to 9.05 Mt $CO_2$-eq (−40% to +74%). The effluent $N_2O$ emission factor changed them by −2.5% to +40%, the MCF for aerobic treatment by −11% to +24%, and the grid emission factor by ±3.2%. Surplus biogas matters more on this basis than on the plant mean, because digesters are concentrated at large plants. If 99% of the surplus biogas were combusted, the annual gross emissions would fall by 7.2%. The $N_2O$ emission factors measured at Korean plants (0.94–26.73 g $N_2O$/kg TN [23]) lie within the IPCC range applied here (0.25–70.7 g $N_2O$/kg TN). For the $N_2O$ emission factor for biological treatment, the aerobic MCF, and the effluent $N_2O$ emission factor, the upward swing in

Figure 8 is larger than the downward swing because the IPCC uncertainty ranges extend further above the default values than below them (Supplementary Table S4); the possible deviation of the results is therefore larger on the upper side. The parameters were varied one at a time, so the combined uncertainty was not quantified. The IPCC ranges give bounds but not probability distributions, so we did not derive a probabilistic interval. The analysis therefore shows the influence of the selected parameters one by one and does not cover interactions between them. It also does not cover the uncertainty from incomplete activity data, such as the sludge without an identified final route and the activities that were not reported. Since the absolute values depend heavily on the choice of $N_2O$ emission factor, developing measurement-based $N_2O$ emission factors for Korean plants would be the most effective way to improve the accuracy of operational emission estimates. This study is also a cross-sectional analysis of a single year (2024) of statistics in which the same Tier 1 factors were applied to every plant, and intermediately treated sludge without an identified final route (about 45% of the generated amount) and flare operation could not be included in the assessment; these limitations could be addressed as plant-level measured data become available. In addition, we compared the emissions per cubic meter of treated effluent. This unit does not reflect the differences in influent strength and effluent quality among plants. Comparison per unit of BOD or nitrogen removed, or per population equivalent, remains for further study. The excluded sludge is large relative to the sludge assigned to the four routes. Its exclusion affects both the sludge-related emissions and the recovery credits, so both are lower estimates. The direction and size of the effect on the net emissions cannot be established, because the final destinations are unknown. In particular, the final destinations of the intermediately treated sludge, the activities that were not reported, and the moisture content imputed for 120 plants could change the sludge-related emissions and credits, and their effect should be examined when the final disposal routes are reported.

## 4. Conclusions

This study estimated the operational greenhouse gas emissions of 610 public WWTPs in Korea at the plant level from the 2024 Sewerage Statistics. On-site emissions, electricity, downstream sources and conditional resource-recovery credits were evaluated within a single system boundary.

The annual gross emissions of the 610 plants were 5.207 Mt $CO_2$-eq, or 0.723 kg $CO_2$-eq/$m^3$ of effluent on a flow-weighted basis. $N_2O$ from treatment and receiving water was the largest source (43.5%), followed by electricity (31.8%). Plants of ≥100,000 $m^3$/d produced 67.3% of the total. The arithmetic plant mean was higher, at 1.039 kg $CO_2$-eq/$m^3$, and electricity was its largest component (53.2%). The difference between the two measures came almost entirely from electricity. The electricity intensity fell from 0.687 kg $CO_2$-eq/$m^3$ in the 500–10,000 $m^3$/d class to 0.212 kg $CO_2$-eq/$m^3$ in the ≥100,000 $m^3$/d class. Among the process groups, the plant-mean intensity ranged from 0.696 (AS–biofilm–BNR) to 1.495 kg $CO_2$-eq/$m^3$ (MBR). These values rest on uniform emission factors. Some differences remained after adjustment for capacity and influent strength, but they do not establish causal process effects. The conditional credits for effluent reuse, biogas use and sludge recovery were 0.457 Mt $CO_2$-eq, or 8.8% of the gross emissions. Net emissions were 4.750 Mt $CO_2$-eq. Plants with no resource recovery had no credit. Plants with all three activities reduced their gross intensity by 12.3% on average.

These results suggest that energy efficiency, $N_2O$ control in nitrogen removal, and resource recovery are the main options for reducing the emissions of WWTPs. Their effects were not tested with mitigation scenarios in this study and need to be evaluated for individual plants. Anaerobic digestion was installed at only 10.2% of the plants. At the plants reporting biogas production, $CH_4$ from surplus biogas averaged 0.125 kg $CO_2$-eq/$m^3$

when all surplus was assumed to be vented. This exceeded the biogas credit (0.033 kg $CO_2$-eq/$m^3$). The benefit of expanding digesters therefore depends on flaring or using the surplus biogas. The priority is also likely to differ with plant size. Small plants have the highest unit emissions, and electricity efficiency matters most there. Large plants account for most of the total, and $N_2O$ and electricity matter most there. As the grid emission factor decreases, the share of electricity will fall and that of $N_2O$ will increase. These values depend strongly on the $N_2O$ emission factor for treatment, and measurement-based factors for Korean plants are needed to reduce this uncertainty.

**Supplementary Materials:** The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/w18232965/s1, Table S1: Classification of the reported treatment processes into five process families; Table S2: Reporting status of the variables and treatment of missing values (610 plants); Table S3: Plant-level emission intensity (kg $CO_2$-eq/$m^3$) by (a) treatment process group and (b) capacity class; Table S4: Parameters, ranges and results of the one-at-a-time sensitivity analysis; Table S5: Mean emission intensity by emission source and treatment process group (kg $CO_2$-eq/$m^3$; values of Figures 4 and 5); Table S6: Mean credits, net emission intensity and reduction rate by treatment process group (kg $CO_2$-eq/$m^3$; values of Figure 6); Table S7: Emission intensity of the plants with anaerobic digestion and flare scenarios; Table S8: Influent and effluent T-N and T-N removal by treatment process group; Table S9: Regression of the gross emission intensity and nonparametric tests among the process groups; Table S10: Comparison with the 2024 national inventory for domestic wastewater.

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**Data Availability Statement:** The plant-level data analyzed in this study are derived from the 2024 Sewerage Statistics of the Ministry of Environment, Republic of Korea, which are publicly available at https://www.hasudoinfo.or.kr/ (accessed on 29 July 2026). Summary statistics are provided in the Supplementary Materials; plant-level results are available from the corresponding author on reasonable request.

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## References

1. IPCC. *Climate Change 2022: Mitigation of Climate Change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change—Summary for Policymakers*; Cambridge University Press: Cambridge, UK, 2022. [CrossRef]
2. Greenhouse Gas Inventory and Research Center (GIR). National GHG Inventory Management System. Available online: https://www.gir.go.kr/eng/index.do?menuId=9 (accessed on 29 July 2026).
3. Gu, Y.; Li, Y.; Yuan, F.; Yang, Q. Optimization and control strategies of aeration in WWTPs: A review. *J. Clean. Prod.* **2023**, *418*, 138008. [CrossRef]
4. Ministry of Environment. 2024 Statistics of Sewerage; Sewerage Information System. Available online: https://www.hasudoinfo.or.kr/ (accessed on 29 July 2026). (In Korean)
5. Korea Power Exchange. *2024 Electricity Market Statistics*; Korea Power Exchange: Naju, Republic of Korea, 2025. Available online: https://www.kpx.or.kr/board.es?mid=a10502000000&bid=0045&list_no=74718&act=view (accessed on 22 October 2026). (In Korean)

6. World Resources Institute (WRI); World Business Council for Sustainable Development (WBCSD). *The Greenhouse Gas Protocol: A Corporate Accounting and Reporting Standard*, Revised ed.; WRI: Washington, DC, USA; WBCSD: Geneva, Switzerland, 2004.
7. Huang, L.; Li, H.; Li, Y. Greenhouse gas accounting methodologies for wastewater treatment plants: A review. *J. Clean. Prod.* **2024**, *448*, 141424. [CrossRef]
8. Anh, L.T.L.; Le Luu, T. Greenhouse gas emissions from municipal wastewater treatment: Global insights and Vietnam's approach. *Curr. Opin. Environ. Sci. Health* **2025**, *47*, 100655. [CrossRef]
9. Bao, Z.; Sun, S.; Sun, D. Characteristics of direct $CO_2$ emissions in four full-scale wastewater treatment plants. *Desalin. Water Treat.* **2015**, *54*, 1070–1079. [CrossRef]
10. Appels, L.; Baeyens, J.; Degrève, J.; Dewil, R. Principles and potential of the anaerobic digestion of waste-activated sludge. *Prog. Energy Combust. Sci.* **2008**, *34*, 755–781. [CrossRef]
11. Kampschreur, M.J.; Temmink, H.; Kleerebezem, R.; Jetten, M.S.M.; van Loosdrecht, M.C.M. Nitrous oxide emission during wastewater treatment. *Water Res.* **2009**, *43*, 4093–4103. [CrossRef] [PubMed]
12. Li, L.; Lee, G.; Kang, D. Energy consumption and greenhouse gas (GHG) emissions in urban wastewater treatment facilities: A case study of Seoul Metropolitan City (SMC). *Water* **2025**, *17*, 464. [CrossRef]
13. Szulc, P.; Kasprzak, J.; Dymaczewski, Z.; Kurczewski, P. Life cycle assessment of municipal wastewater treatment processes regarding energy production from the sludge line. *Energies* **2021**, *14*, 356. [CrossRef]
14. Bhowmik, P.; Ahmed, T. Environmental impact of a domestic wastewater treatment plant using LCA approach under various treatment scenarios: A case study in Bangladesh. *Water Sci. Technol.* **2026**, *93*, 534–551. [CrossRef] [PubMed]
15. Tsangas, M.; Papamichael, I.; Banti, D.; Samaras, P.; Zorpas, A.A. LCA of municipal wastewater treatment. *Chemosphere* **2023**, *341*, 139952. [CrossRef] [PubMed]
16. Mannina, G.; Ekama, G.; Caniani, D.; Cosenza, A.; Esposito, G.; Gori, R.; Garrido-Baserba, M.; Rosso, D.; Olsson, G. Greenhouse gases from wastewater treatment—A review of modelling tools. *Sci. Total Environ.* **2016**, *551–552*, 254–270. [CrossRef] [PubMed]
17. Duque, A.F.; Campo, R.; Val del Rio, A.; Amorim, C.L. Wastewater valorization: Practice around the world at pilot- and full-scale. *Int. J. Environ. Res. Public Health* **2021**, *18*, 9466. [CrossRef] [PubMed]
18. Verstraete, W.; Van de Caveye, P.; Diamantis, V. Maximum use of resources present in domestic "used water". *Bioresour. Technol.* **2009**, *100*, 5537–5545. [CrossRef] [PubMed]
19. El Abbadi, S.H.; Feng, J.; Hodson, A.R.; Amouamouha, M.; Busse, M.; Polcuch, C.; Zhou, P.; Macknick, J.; Guest, J.S.; Stokes-Draut, J.; et al. Benchmarking greenhouse gas emissions from US wastewater treatment for targeted reduction. *Nat. Water* **2025**, *3*, 1133–1143. [CrossRef]
20. Chen, J.; Ao, Z.; Chen, H.; Wang, Y.; Jiang, M.; Qi, L.; Liu, G.; Wang, H. Analyzing greenhouse gas emissions and influencing factors of 247 actual wastewater treatment plants in China using emission factor and operational data integrated methods (ODIM). *Environ. Sci. Pollut. Res.* **2024**, *31*, 37387–37403. [CrossRef] [PubMed]
21. Hwang, K.-L.; Bang, C.-H.; Zoh, K.-D. Characteristics of methane and nitrous oxide emissions from the wastewater treatment plant. *Bioresour. Technol.* **2016**, *214*, 881–884. [CrossRef] [PubMed]
22. Kim, S.-B.; Yoon, S.-Y.; Lee, Y.-K.; Lee, S.-G.; Lee, S. Development of greenhouse gas emission factor in wastewater treatment section in Korea. In *Proceedings of the 17th Workshop on Greenhouse Gas Inventories in Asia (WGIA17), Singapore, 30 July–2 August 2019*; Center for Global Environmental Research, National Institute for Environmental Studies: Tsukuba, Japan, 2019; p. 35. Available online: https://www.nies.go.jp/gio/en/jqjm1000000kbl6r-att/web_CGER-I147-2019_WGIA17.pdf (accessed on 19 November 2026).
23. Yang, H.-J.; Park, J.-M.; Kim, M.-J. Estimate of nitrous oxide emission factors from municipal wastewater treatment plants. *J. Korean Soc. Environ. Eng.* **2008**, *30*, 1281–1286. (In Korean)
24. Ministry of Climate, Energy and Environment. *Guidelines for Establishing Comprehensive Sewerage Plans*; Ministry of Climate, Energy and Environment: Sejong, Republic of Korea, 2025. Available online: https://mcee.go.kr/home/web/policy_data/read.do?condition.deleteYn=N&condition.orderSeqId=8457&condition.rnSeq=171&maxIndexPages=10&maxPageItems=10&menuId=92&pagerOffset=0&seq=8458 (accessed on 29 July 2026). (In Korean)
25. Oh, J.R.; Kim, H.S.; Kim, D.W.; Phae, C.G.; Lee, J.Y. A study on generation characteristics and production unit of sewage waste according to sewage treatment requirements. *J. Korea Soc. Waste Manag.* **2019**, *36*, 777–786. (In Korean) [CrossRef]
26. Greenhouse Gas Protocol. *IPCC Global Warming Potential Values, Version 2.0*; World Resources Institute (WRI) and World Business Council for Sustainable Development (WBCSD): Washington, DC, USA, 2024.
27. IPCC. Waste, Chapter 6: Wastewater Treatment and Discharge. In *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IPCC: Geneva, Switzerland, 2019; Volume 5.
28. Ministry of Climate, Energy and Environment. Enforcement Rule of the Sewerage Act, Annex 1: Effluent Quality Standards for Public Sewage Treatment Facilities. Available online: https://www.law.go.kr/ (accessed on 19 November 2026). (In Korean)
29. UNFCCC. *Clean Development Mechanism Methodological Tool 14: Project and Leakage Emissions from Anaerobic Digesters, Version 03.0*; United Nations Framework Convention on Climate Change: Bonn, Germany, 2024.

30. IPCC. Waste, Chapter 5: Incineration and Open Burning of Waste. In *2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IGES: Hayama, Japan, 2006; Volume 5.
31. IPCC. *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories*; IPCC: Hayama, Japan, 2000.
32. IPCC. Waste, Chapter 2: Waste Generation, Composition, and Management Data. In *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IPCC: Geneva, Switzerland, 2019; Volume 5.
33. IPCC. Waste, Chapter 4: Biological Treatment of Solid Waste. In *2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IGES: Hayama, Japan, 2006; Volume 5.
34. Yu, S.; Deng, S.; Zhou, A.; Wang, X.; Tan, H. Life cycle assessment of energy consumption and GHG emission for sewage sludge treatment and disposal: A review. *Front. Energy Res.* **2023**, *11*, 1123972. [CrossRef]
35. IPCC. Energy, Chapter 2: Stationary Combustion. In *2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IGES: Hayama, Japan, 2006; Volume 2.
36. Korea Environmental Industry & Technology Institute (KEITI). Environmental Product Declaration (EPD) Characterization Factors. Available online: https://ecosq.or.kr/websquare.do#w2xPath=/cm/main/index.xml (accessed on 24 November 2026).
37. IPCC. Energy, Chapter 1: Introduction. In *2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IGES: Hayama, Japan, 2006; Volume 2.
38. Tappen, S.J.; Aschmann, V.; Effenberger, M. Lifetime development and load response of the electrical efficiency of biogas-driven cogeneration units. *Renew. Energy* **2017**, *114*, 857–865. [CrossRef]
39. Terhan, M.; Comakli, K. Energy and exergy analyses of natural gas-fired boilers in a district heating system. *Appl. Therm. Eng.* **2017**, *121*, 380–387. [CrossRef]
40. Georgiopoulou, M.; Lyberatos, G. Life cycle assessment of the use of alternative fuels in cement kilns: A case study. *J. Environ. Manag.* **2018**, *216*, 224–234. [CrossRef] [PubMed]
41. Mateus, M.M.; Neuparth, T.; Cecílio, D.M. Modern kiln burner technology in the current energy climate: Pushing the limits of alternative fuel substitution. *Fire* **2023**, *6*, 74. [CrossRef]
42. Park, S.W.; Jang, C.H. Fuel characteristics of sewage sludge as alternative fuel of coal. *J. Korea Soc. Waste Manag.* **2010**, *27*, 431–437. (In Korean)
43. Keene, N.A.; Reusser, S.R.; Scarborough, M.J.; Grooms, A.L.; Seib, M.; Santo Domingo, J.; Noguera, D.R. Pilot plant demonstration of stable and efficient high rate biological nutrient removal with low dissolved oxygen conditions. *Water Res.* **2017**, *121*, 72–85. [CrossRef] [PubMed]
44. Drews, A. Membrane fouling in membrane bioreactors—Characterisation, contradictions, cause and cures. *J. Membr. Sci.* **2010**, *363*, 1–28. [CrossRef]
45. Askari, S.S.; Giri, B.S.; Basheer, F.; Izhar, T.; Ahmad, S.A.; Mumtaz, N. Enhancing sequencing batch reactors for efficient wastewater treatment across diverse applications: A comprehensive review. *Environ. Res.* **2024**, *260*, 119656. [CrossRef] [PubMed]
46. Rodriguez-Caballero, A.; Aymerich, I.; Marques, R.; Poch, M.; Pijuan, M. Minimizing $N_2O$ emissions and carbon footprint on a full-scale activated sludge sequencing batch reactor. *Water Res.* **2015**, *71*, 1–10. [CrossRef] [PubMed]
47. Saini, S.; Tewari, S.; Dwivedi, J.; Sharma, V. Biofilm-mediated wastewater treatment: A comprehensive review. *Mater. Adv.* **2023**, *4*, 1415–1443. [CrossRef]
48. He, Y.; Liu, Y.; Li, X.; Zhu, T.; Liu, Y. Unveiling the roles of biofilm in reducing $N_2O$ emission in a nitrifying integrated fixed-film activated sludge (IFAS) system. *Water Res.* **2023**, *243*, 120326. [CrossRef] [PubMed]
49. Kaminski, G.F.; Mannich, M.; Possetti, G.R.C. In situ efficiency evaluation of enclosed flares for biogas burning in sewage treatment plants. *Eng. Sanit. Ambient.* **2025**, *30*, e20240134. [CrossRef]
50. UNFCCC. *Clean Development Mechanism Methodological Tool 06: Project Emissions from Flaring, Version 04.0*; United Nations Framework Convention on Climate Change: Bonn, Germany, 2022. Available online: https://cdm.unfccc.int/methodologies/PAmethodologies/tools/am-tool-06-v4.0.pdf (accessed on 19 November 2026).
51. Volschan Junior, I.; de Almeida, R.; Cammarota, M.C. A review of sludge pretreatment methods and co-digestion to boost biogas production and energy self-sufficiency in wastewater treatment plants. *J. Water Process Eng.* **2021**, *40*, 101857. [CrossRef]
52. Lima, D.; Appleby, G.; Li, L. A scoping review of options for increasing biogas production from sewage sludge: Challenges and opportunities for enhancing energy self-sufficiency in wastewater treatment plants. *Energies* **2023**, *16*, 2369. [CrossRef]
53. Greenhouse Gas Inventory and Research Center (GIR). *2026 National Greenhouse Gas Inventory (1990–2024)*; GIR: Cheongju, Republic of Korea, 2026. Available online: https://www.gir.go.kr/home/index.do?menuId=36 (accessed on 19 November 2026). (In Korean)


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