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Statistical Analysis of Global Warming Potential, Eutrophication Potential, and Sludge Production of Wastewater Treatment Plants in Japan

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Abstract: Treatment performance and greenhouse gas emissions of various biological sewage treatment processes in Japan were evaluated. Data related to energy consumption, effluent water quality, and sludge production of various treatment processes were obtained from "Sewage Statistics" published by the Japan Sewage Works Association. The conventional activated sludge (CAS) process and modified processes for nutrient removal were selected for analysis, such as anaerobic–oxic, recycled nitrification–denitrification, anaerobic-anoxic–oxic, nitrification/endogenous denitrification, and step-feed nitrification–denitrification processes. Performance of the treatment processes was evaluated as the eutrophication potential (EP) calculated from the BOD, total nitrogen, and total phosphorus concentrations in treated effluent and NO_X emission by electricity consumption. The global warming potential (GWP) of treatment processes was calculated from CO_2, CH_4, and N_2O emissions by electricity consumption, and N_2O and CH_4 emissions from the biological processes and water environment where effluent is discharged. The EP values of the nutrient removal processes (1.0–6.4 g-PO_4eq/m^3) showed negative correlation with the GWP values (0.22–0.68 kg-CO_2eq/m^3) as a general trend. The sole exception to this tradeoff was the step-feed nitrification–denitrification process, which can reduce the EP value of sewage with a considerably smaller increase of the GWP value than that of the CAS process. Sludge yields of treatment processes (0.01–0.032 m^3/m^3) also showed negative correlation with GWP values.

Keywords: Eutrophication potential, global warming potential, life cycle assessment, sludge production, wastewater treatment plant.

1. Introduction

In Japan, the conventional activated sludge (CAS) process, mainly used for biochemical oxygen demand (BOD) removal, has been the mainstream method used at wastewater treatment plants (WWTPs) since 1930. However, eutrophication in closed water bodies has reduced water quality drastically during the rapid economic growth period because the CAS process cannot remove nutrients from sewage efficiently. In 1978, the total Pollutant Load Regulation set allowable limits of loading rates for chemical oxygen demand, total nitrogen (T–N), and total phosphorus (T–P) for industrial plants. The Ministry of the Environment also set the effluent standard for T–N and T–P in 1985 to prevent lakes and marshes from eutrophication. The Water Pollution Control Law was amended in 1993 to improve water quality in eutrophicated ocean areas, lakes, and marshes. The number of WWTPs with activated sludge processes modified for efficient nutrient removal has been increasing since amendment of the Law [1].

Advanced treatment processes for nutrient removal reduce the eutrophication impact on the water environment, but they are expected to necessitate higher energy consumption than the CAS process. Little attention has been devoted to this tradeoff because the benefit from the improvement of water quality is believed to be greater than the environmental impact derived from energy consumption. However, worldwide efforts for reduction of the greenhouse gas (GHG) emissions have become an urgent concern, especially since the Kyoto Protocol was adopted in 1997. Several studies have assessed environmental impacts of WWTPs [2-4], including those in Japan [5-6]. In the operational stage of WWTPs, large amounts of electricity are consumed mainly for pumping air into aeration tanks to activate microorganisms. Moreover, along with CO_2 emissions that occur with electricity consumption, N_2O and CH_4 are emitted as GHGs from biological treatment processes [7]. Vidal et al. [4] reported that the oxidation ditch (OD) process would be superior to the Ludzack–Ettinger process as a nitrogen removal process in Spain, taking all environmental impacts together, such as global warming and eutrophication. Although the GHG emissions are expected to depend on configurations and the operational conditions, only a few studies have compared different biological treatment processes from the perspectives of both water quality and GHG emissions.

However, the quantity of sewage sludge generated from the biological treatment processes has increased apace with sewerage development. Typically in Japan, sewage sludge is incinerated after being thickened and dewatered with a large amount of energy consumption and GHG emissions. Methane emitted from anaerobic digestion treatment has been used to heat digestion tanks and has been used as a supplementary fuel for incinerators. Moreover, power generation using digestion gas is progressing as a substitute for fossil fuels. Although the goals of sewage sludge treatment have been stabilization and volume reduction, sewage sludge is being reconsidered for use as biomass for energy utilization, for use as construction material, and as other resources [8]. Sludge treatment should also be considered for total evaluation of environmental impacts derived from wastewater treatment processes.

In this study, the treatment performance, the GHG emissions, and sludge production of various sewage treatment technologies were evaluated comparatively based on statistical data of WWTPs in Japan. Our assessment of the value of WWTPs will be changed by reviewing the performance and GHG emissions for sustainable sewage treatment.

2. Experimental

2.1 Data collection and process configurations of WWTPs in Japan

From data of sewage work in Japan [9], data related to energy consumption, water quality, and waste sludge production were collected. Now in Japan, there are about 1500 WWTPs with various configurations. The classification and numbers of biological treatment processes are presented in Table 1. They have different configurations and the planned effluent quality for BOD, T–N, and T–P. The schematic flows of the treatment processes for nutrient removal (anaerobic–oxic (AO), recycled nitrification–denitrification, anaerobic–anoxic–oxic (A2O), nitrification/endogenous denitrification, and step-feed nitrification–denitrification processes) and other purposes (step-aeration, oxygen aeration, extended aeration, and oxidation ditch processes) are presented, respectively, in Figs. 1 and 2.

The CAS process apparatus includes a primary settling tank (PST), an aeration tank (AT) for BOD removal, and a secondary settling tank (SST) for sludge thickening. The typical sludge retention time (SRT) and hydraulic retention time (HRT) of the CAS process are, respectively, 7–15 d and 6–8 h. The AO process consists of an anaerobic (anoxic) tank followed by an oxic tank. Phosphorus is released from bacterial cells to wastewater in the anaerobic tank. Subsequently, the bacteria uptake phosphorus in the oxic tank excessively. This process is also effective to prevent bulking. The recycled nitrification–denitrification process consists of an anoxic tank followed by an oxic tank with internal recirculation from the oxic tank to the anoxic tank. Organic nitrogen is decomposed by heterotrophs to ammonia–nitrogen. Subsequently, the ammonia–nitrogen is oxidized into nitrate via nitrite by autotrophic bacteria. In anoxic conditions, nitrate is reduced to N_2 with oxidation of BOD. In a manner of speaking, the A2O process is a hybrid of the AO process and the recycled nitrification–denitrification process. Through the combination of anaerobic, anoxic, and oxic tanks in series with internal recirculation from the oxic tank to the anoxic tank, both nitrogen and phosphorus are removed. The nitrification – endogenous denitrification process consists of an oxic tank followed by an anoxic tank. Nitrate produced in the oxic tank is reduced by denitrifying bacteria using internal storage products in the anoxic tank. Long HRT operation (16–24 h) is necessary for the endogenous denitrification. The step-feed nitrification–denitrification process consists of configuration of anoxic and oxic conditions in tanks in series and receives influent as an external carbon source for denitrification in the anoxic tanks.

In the step aeration process, influent branches along the tank length. This process results in a more uniform oxygen supply in the tank and a more stable environment for microorganisms. The oxygen aeration process supplies pure oxygen (>95%) to the aeration tank instead of air. Although much electricity is consumed in the oxygen generation process, the sludge concentrations can be approximately doubled and HRT can be reduced to half that of the CAS process through highly dissolved oxygen concentration. In the extended aeration process, sludge is retained in the aeration tank until the production rate of new cells equals the decay rate of existing cells for achieving little excess sludge production. This process is usually not equipped with the PST and is suitable for small WWTPs because of long HRT (16–24 h). In the OD process, a circular aeration basin is used, with rotary brush aerators that extend across the ditch width. Brush aerators keep the sludge in suspension and drive the wastewater around the circular channel. This process is also suitable for small WWTPs because of the resultant long HRT (24–48 h). The OD process without the PST is popular in small WWTPs (Table 1).

2.2 Evaluation of environmental impacts

The study domain was delimited to evaluate the impacts produced solely by operation of the WWTPs. No consideration was given to the energy and natural resources necessary to build the WWTP facilities. This assumption was based on reports that the environmental impacts of the construction phase of a WWTP do not differ much among different configurations [2].

Software (JEMAI-LCA Pro; Japan Environmental Management Association for Industry, Tokyo) was used to estimate the environmental impact of the sewage treatment processes. Among the available methodologies, a Life cycle Impact assessment Method based on Endpoint modeling (LIME) [11] was used to estimate eutrophication and global warming impacts. The eutrophication potential (EP) for treating 1.0 m^3 wastewater was calculated from NO_x emissions in electricity consumption and BOD, T–N, and T–P concentrations in effluent. The global warming potential (GWP) for treating wastewater of 1.0 m^3 was calculated from CO_2, CH_4, N_2O, and NO_x emissions in electricity consumption, and CH_4 and N_2O emissions occurring from the biological treatment processes and receiving waters of effluent. The influent characteristics were assumed to be equal for all biological treatment processes.

Weighting factors of NOx, T–N, T–P, and BOD for EP estimation were assumed, respectively, as 0.0011, 0.26, 3.06, and 0.00148 kg-PO_4eq [11]. Weighting factors of CO_2, CH_4, and N_2O for GWP estimation were assumed, respectively, as 1.0, 21, and 310 (kg-CO_2eq) [11]. Emission intensities of CO_2, CH_4, N_2O, and NO_x for 1.0 kWh of electricity consumption were assumed, respectively, as typical values, 0.446 kg-CO_2, 9.45×10^{-6} kg-CH_4, 1.94×10^{-5} kg-N_2O, and 0.239 kg-N_2O [11]. The emission intensities of CH_4 and N_2O from the biological treatment processes used for this study are shown respectively in Tables 2 and 3. Those values are expected to depend on biological treatment processes, but detailed data for most processes are unavailable at present. Emission intensities of N_2O and CH_4 in receiving waters of effluent were assumed, respectively, as 6.0×10^{-2} kg-CH_4/kg-BOD and 7.9×10^{-3} kg-N_2O/kg-N [18].

Table 1. Classification and planned effluent quality of biological wastewater treatment processes.

	Number of WWTPs [a]			Planned effluent quality, mg/L [b]		
	Annual treatment, m^3/y			(Typical removal, %)		
	$< 10^6$	$10^6–10^8$	$> 10^8$	BOD	T-N	T-P
Conventional	63	282	145	10–15 (90–95)		
Anaerobic-oxic	14	10	15	10–15		< 3 (75–95)
Recycled nitrification–denitrification	7	6	0	10–15	< 20 (65–75)	
Anaerobic-anoxic-oxic	5	4	2	10–15	< 20 (65–75)	< 3 (75–95)
Nitrification/ endogenous denitrification	4	1	0		(75–95)	
Step-feed nitrification–denitrification	0	3	2		(75–85)	
Step aeration	0	4	6	10–15 (90–95)		
Oxygen aeration	1	3	3	(90–95)		
Extended aeration	25	1	0	(90–95)		
Oxidation ditch	459	30	0	(90–95)		

[a] Japan Sewage Works Association [9]. [b] Japan Sewage Works Association [10].

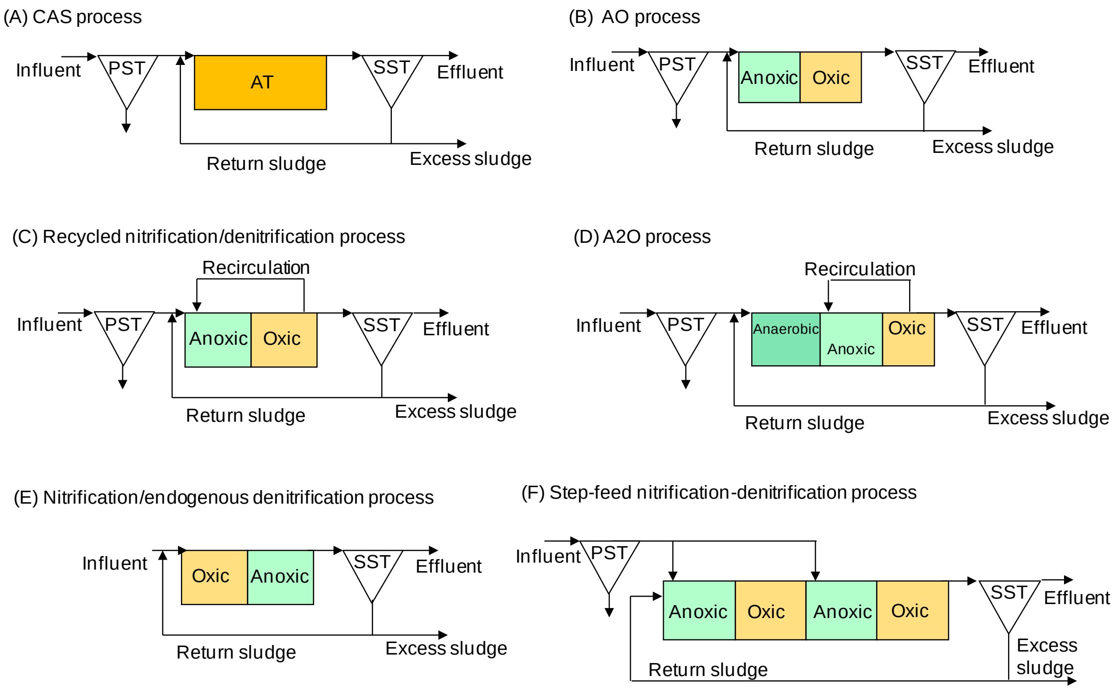

Figure 1. Variations of biological sewage treatment processes for nutrient removal: (A) conventional activated sludge (CAS) process, (B) anaerobic–oxic (AO) process, (C) recycled nitrification–denitrification process, (D) anaerobic–anoxic–oxic (A2O) process, (E) nitrification/ endogenous denitrification process, and (F) step-feed nitrification–denitrification process. PST and SST respectively represent primary and secondary settling tanks.

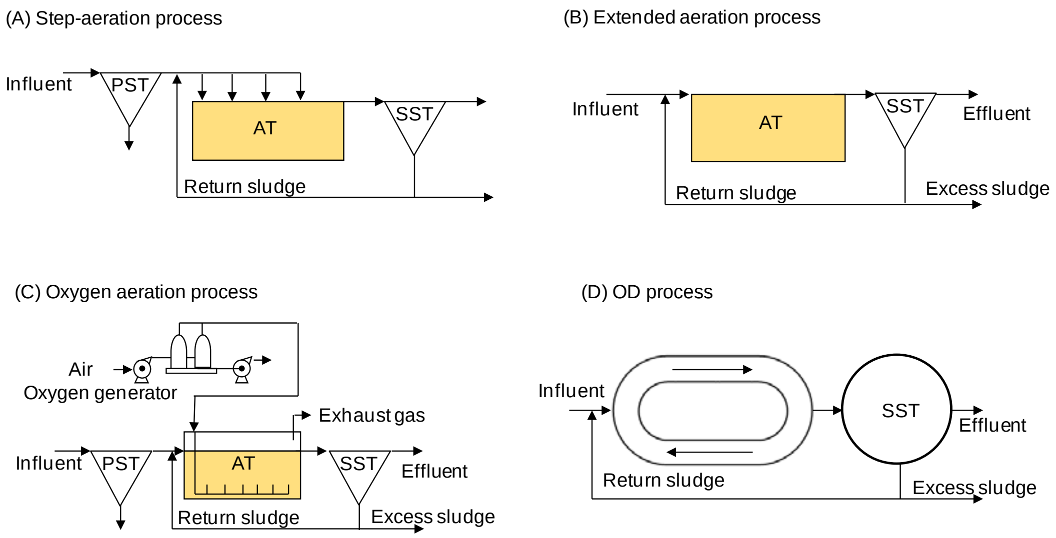

Figure 2. Variations of biological sewage treatment processes: (A) step-aeration process, (B) oxygen aeration process, (C) extended aeration process, and (D) oxidation ditch (OD) process.

Table 2. Emission intensity of CH_4 from biological treatment processes (mg-CH_4/m^3, total from settling tanks, biological reaction tanks, and secondary treatment tanks).

Process	Intensity	Remarks
Conventional	5.3×10^{-4}	Average of 14 samples from 8 WWTs [12-15]
Anaerobic-oxic	2.6×10^{-4}	Average of 2 samples from 1 WWTP [15]
Recycled nitrification–denitrification	2.4×10^{-4}	1 sample [12]
Anaerobic-anoxic-oxic	1.8×10^{-4}	1/3 of the CAS process [16]
Others	5.3×10^{-4}	No data. The CAS process value was used.

Table 3. Emission intensity of N_2O from biological treatment processes (mg-N_2O/m^3, total from settling tanks, biological reaction tanks, and secondary treatment tanks).

Process	Intensity	Remarks
Conventional	1.6×10^{-4}	Average of 7 samples from 4 WWTPs [14-15, 17]
Anaerobic-oxic	6.1×10^{-5}	1 sample [15]
Anaerobic-anoxic-oxic	2.0×10^{-5}	1/8 of the CAS process [16]
Others	1.6×10^{-4}	No data. The CAS process value was used.

3. Results

3.1 Eutrophication potential of biological sewage treatment processes

Average concentrations of BOD, T–N, and T–P in sewage were, respectively, 180 mg/L, 34 mg/L, and 4.1 mg/L (*n* = 617). All treatment processes were able to reduce the BOD concentration to < 5 mg/L. However, the processes differed in terms of the nitrogen and phosphorus emissions through plant effluent. The EP values estimated for the selected biological treatment processes are depicted in Fig. 3. The EP value of untreated sewage was estimated as 22 g-PO_4eq/m^3. The average T–N and T–P concentrations in effluent of the CAS process were, respectively, 15 mg/L and 1.0 mg/L according to data of sewage work in Japan [9]. The EP value derived from effluent of the CAS process was estimated as 6.4 g-PO_4eq/m^3, indicating its great reduction compared with the direct discharge of raw sewage. The nutrient removal processes were regarded as lower EP values to < 4 g-PO_4eq/m^3. Among the compared processes, the nitrification/endogenous denitrification process showed the smallest EP value (1.6 g-PO_4eq/m^3).

The estimated EP values of other modified processes, such as the step-aeration, oxygen aeration, extended aeration, and the OD processes, were as high as the CAS process. The advantages of those processes cannot be highlighted by the term of the eutrophication.

3.2 Global warming potential of biological sewage treatment processes

The estimated GWP values of the selected biological treatment processes are portrayed in Fig. 4. Untreated sewage was inferred to have the GWP value of 0.31 kg-CO_2eq/m^3, mainly attributable to CH_4 and N_2O emissions in water environment where sewage is discharged. In a water environment, CH_4 is produced by archaea under anaerobic conditions. N_2O is also produced by nitrifying bacteria under oxic conditions and denitrifying bacteria under anoxic conditions. The CH_4 and N_2O emission factors engender large uncertainty because related microbial reactions depend highly on environmental conditions such as temperature, pH, and dissolved oxygen concentrations.

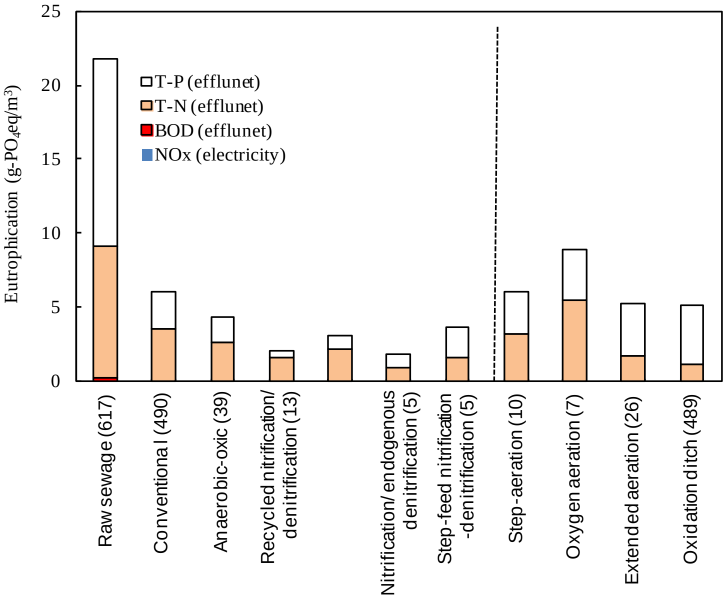

Figure 3. Eutrophication potential of various biological treatment processes in WWTPs. Sample numbers are shown in parentheses.

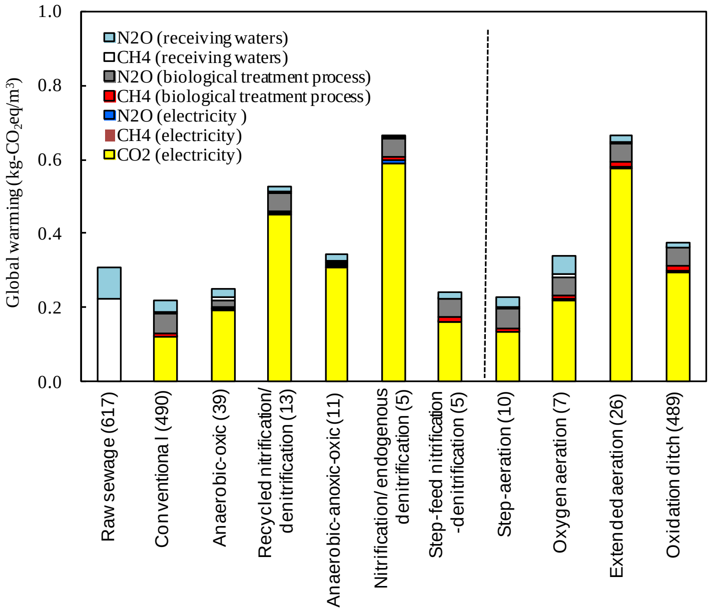

Figure 4. Global warming potential of various biological treatment processes in WWTPs. Sample numbers are shown in parentheses.

The CAS process was estimated as reducing the GWP value to 0.22 kg-CO_2eq/m^3. The GWP caused by the CAS process consists mainly of CO_2 derived from electricity consumption (54%), N_2O in the biological treatment process (23%), and N_2O in water environment where the effluent is discharged (15%). The CO_2 emission intensity of electricity consumption used for this study was a typical value (0.446 kg-CO_2/kWh) but locally it is 0.33–0.48 kg-CO_2/kWh, depending on the power sources of respective power companies in Japan. Some treatment processes showed higher GWP values than those of untreated sewage. Especially, the recycled nitrification–denitrification, the nitrification/ endogenous denitrification, and the extended aeration processes showed high GWP values attributable to the large consumption of electricity. Modified processes for nitrogen removal commonly need longer SRT operations and sufficient oxygen supply for maintaining nitrifying bacteria. Consequently, the recycled nitrification–denitrification process requires large amounts of electricity for recirculation of the nitrified liquor. The nitrification – endogenous denitrification and the extended aeration processes consume much electricity for aeration under longer HRT operations. Furthermore, the electricity utilization efficiency of the nitrification/endogenous denitrification process is expected to be low because this process has been generally installed in small WWTPs (Table 1). Reportedly, the specific energy consumption (kWh/m^3) in WWTPs decreases inversely with the annual sewage quantity because of the merits of scale [19].

It is noteworthy that the contribution to GHGs, especially N_2O, produced in the biological treatment processes, is high in the estimated GWP value. High N_2O emissions have been reported with accumulation of nitrite in the biological reaction tanks [20]. Nitrite is the substrate for N_2O production both in the nitrification process and the denitrification process. Complete nitrification for nitrogen removal might decrease the nitrite concentration, resulting in low emissions of N_2O (Table 3). However, sewage before entering WWTPs contains CH_4 produced in sewage pipes under anaerobic conditions, and the aeration in the oxic tanks emits CH_4 from sewage to air. Additionally, some parts of WWTPs can produce and emit CH_4 during biological treatment. However, the CH_4 production rate would be low even in the anaerobic tanks in WWTPs. Further studies of the emission intensity of N_2O and CH_4 must be undertaken for precise estimation of GWP values because they are expected to depend not only on the process configurations but also on the operational conditions.

3.3 Sludge production in biological sewage treatment processes

Waste sludge yields in WWTPs are portrayed in Fig. 5. The sludge yield in the PST is principally independent of the biological processes. The median, the first, and the third quantiles of the sludge yield in the PST of 207 CAS, 12 AO, 2 A2O, 3 step-aeration, 3 step-feed nitrification–denitrification, 4 oxygen aeration, 3 recycled nitrification–denitrification and 1 nitrification/endogenous denitrification processes were 0.016, 0.011, and 0.025 m^3/m^3, respectively. The water content of sludge generated in the PST was 99.0–99.5%. In contrast, the sludge yield in the SST of the CAS, the AO, the A2O, and the recycled nitrification–denitrification processes was 0.013–0.017 m^3/m^3. The water content of sludge generated in the SST was 99.2–99.4%. The nitrification/endogenous denitrification, the extended aeration, and the OD processes which have generally no PST, showed low sludge yield. Especially, the extended aeration process, as it was designed, had the lowest sludge yield (0.011 m^3/m^3). Low production of excess sludge in the extended aeration process is expected to accompany low environmental impacts in sludge treatment processes, although its GWP value was high. However, the sludge yield of the step-feed nitrification–denitrification process, which had a low GWP value (Fig. 5) was as high as those of the CAS, the AO, and the A2O processes.

4. Discussion

The relation between the EP and the GWP values of various sewage treatment processes are presented in Fig. 6. Theoretically, the EP and the GWP values are two independent variables. However, except for the step-feed nitrification–denitrification process, the EP values for the nutrient removal processes showed negative correlation with the GWP values. Using regression analysis, the slope of this tradeoff line was estimated as -11.7 g-PO_4eq/kg-CO_2eq. In other words, the reduction of the EP value of 1.0 mg-PO_4eq can increase the GWP of 86.5 g-CO_2eq. This trend is mainly attributed to the large consumption of electricity of the treatment processes for nutrient removal (Fig. 4). It is particularly interesting that the step-feed nitrification–denitrification process was a sole exception to the tradeoff among the nutrient removal processes. Results of this study suggest that this process can reduce the EP values of sewage to 2.1 g-PO_4eq/m^3 with lesser increase in the GWP value. A step-feed anoxic–oxic activated sludge process is an extremely practical method for up-grading of the existing CAS process to enhance nitrogen removal efficiency, eliminating internal water recirculation and supplement of external carbon source for denitrification [21]. Reportedly, a step-feed anoxic–oxic activated sludge process was more economical than the A2O process because the A2O process requires recirculation of nitrified liquor [22]. Consequently, the step-feed nitrification–denitrification process has no recirculation of nitrified liquor, and anoxic tanks in the process consume less electricity than full-aeration tanks. In the past, increased energy consumption or GHG emissions were disregarded as long as the effluent quality was high, but the assessment of its value will change with upgrading and new construction of WWTPs.

The relation between the GWP values and the sludge yields of various sewage treatment processes are presented in Fig. 7. Because the sample numbers were poor except for that of the CAS process, significant differences among sludge yields of the respective treatment processes remained unclear. However, negative correlation was found, unexpectedly, between the GWP values and the sludge yield according to the comparison of the CAS, the extended aeration, the oxygen aeration, the OD, and the step aeration processes. Using regression analysis, the slope of this tradeoff line for those treatment processes was estimated as -0.0455 m^3/kg-CO_2eq or -22.0 kg-CO_2eq/m^3. Strategies for minimization of excess sludge production, such as high dissolved oxygen process, ozonation-combined activated sludge process, control of SRT and biodegradation of sludge in membrane-assisted reactor have been studied [23]. Reportedly, excess sludge production can be reduced by 60% when SRT is increased from 2 to 18 days, but no effect on organic matter removal was observed [24]. The endogenous metabolism under long SRTs can convert substrates to CO_2 and water, resulting in a lower biomass production. It was also reported that the sludge yield in the SST in the pure oxygen aeration process was only 60% of that in the conventional air process [25]. Negative correlation was also found between the GWP values and the sludge yields according to the comparison of the CAS and the nutrient removal processes such as the AO, the A2O, the nitrification/endogenous denitrification processes, the recycled nitrification–denitrification, and the step-feed nitrification and the denitrification processes. Using regression analysis, the slope of this tradeoff line for those treatment processes was estimated as -0.0171 m^3/kg-CO_2eq or -58.4 kg-CO_2eq/m^3.

Vidal et al. [4] also reported environmental impacts of WWTPs with different wastewater treatment configurations.

According to that report, the EP values for the Ludzack-Ettinger process (2.69 kg-PO_4eq) and the OD process (2.09 kg-PO_4eq) were lower than that for the CAS process (8.46 kg-PO_4eq) when treating wastewater at 3800 m^3/d. However, the GWP values for the Ludzack-Ettinger process (207 kg-CO_2eq) and the OD process (202 kg-CO_2eq) were higher than that for the CAS process (142 kg-CO_2eq) [4]. In their LCA assessment boundary, thickening and transportation processes were included with assumptions that the sludge production rates for the Ludzack-Ettinger process (1300 kg/d) and the OD process (1270 kg/d) were lower than that for the CAS process (1500 kg/d) [4]. Those results agree qualitatively with our findings of correlations among the GWP values, the EP values, and the sludge yield of WWTPs, even when considering the differences of wastewater characteristics and boundary conditions.

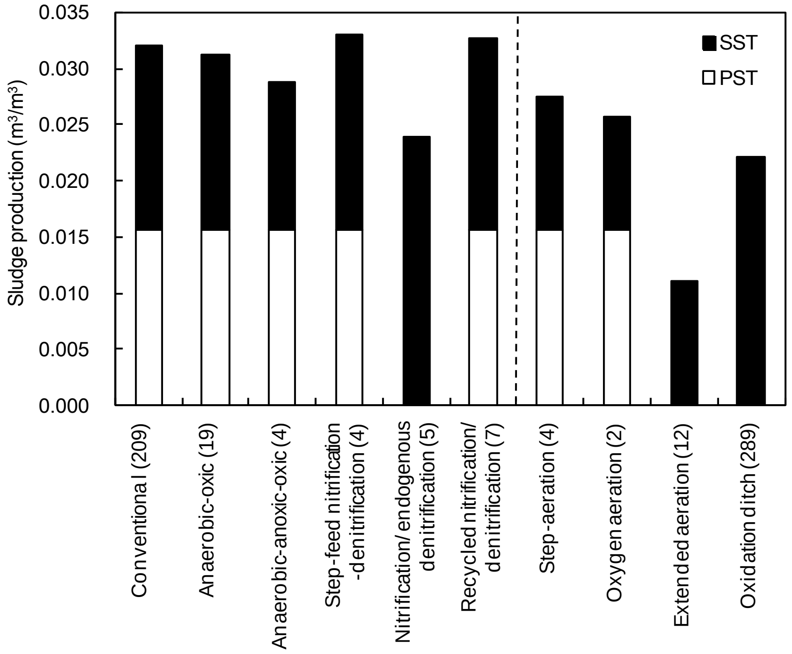

Figure 5. Sludge production from primary settling tank (PST) and secondary settling tank (SST) of various biological treatment processes in WWTPs. Sample numbers of SST are presented in parentheses. The common sludge production value in PST was calculated from the samples of 207 CAS, 12 AO, 2 A2O, 3 step-aeration, 3 step-feed nitrification–denitrification, 4 oxygen aeration, 3 recycled nitrification–denitrification, and 1 nitrification/endogenous denitrification processes.

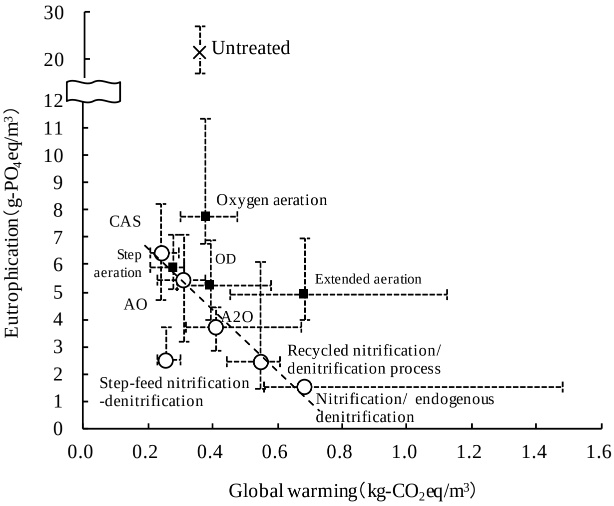

Figure 6. Contributions to eutrophication and global warming of biological wastewater treatment processes in Japan. Open circles and closed squares are respectively represent activated sludge processes modified for nutrient removal and other purposes. Median and 1st–3rd quantiles are shown. Sample numbers are shown in Figs. 3 and 4. The regression equation for CAS, AO, A2O, recycled nitrification–denitrification, and nitrification/endogenous denitrification processes is $y = -0.0865x+0.775$, $r = 0.98$.

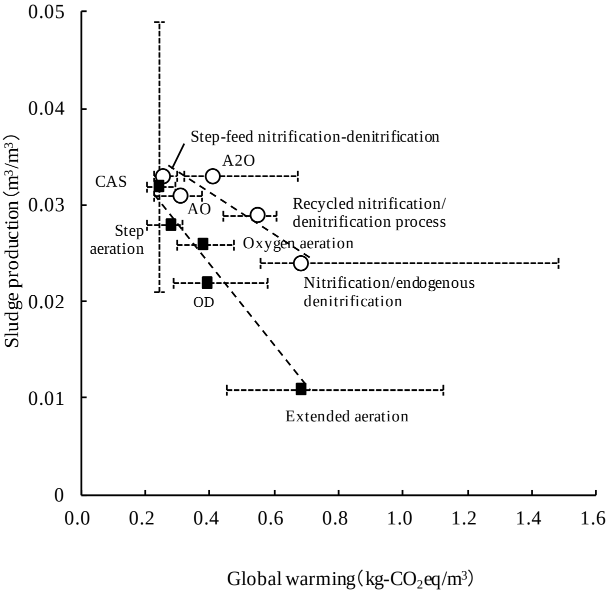

Figure 7. Contributions to sludge production and global warming of biological wastewater treatment processes in Japan. Open circles and closed squares respectively represent activated sludge processes modified for nutrient removal and other purposes. Median and 1st–3rd quantiles are shown for global warming. Sample numbers are presented in Figs. 4 and 5. Regression equations are $y = -0.0171x+0.0375$ ($r = 0.88$) for CAS and nutrient removal processes and $y = -0.0455x+0.0416$ ($r = 0.98$) for CAS and modified processes for other purposes.

Process configurations of sewage sludge treatment plants (SSTPs) are generally independent of those of WWTPs. Now SSTPs in Japan incorporate diverse alternatives of combinations of thickening, anaerobic digestion, composting, dewatering, incineration, melting, and landfill. The energy consumption and the GHG emissions depend on such process configurations and the SSTP scale [26]. Reportedly, sludge treatment processes contributed about 40% of the GHG emissions in WWTPs [8]. The GHG emissions of SSTPs (except for sludge transportation) at sludge loading rates of 2×10^3 and 1.2×10^4 m^3-sludge/d were estimated respectively as 4.0-7.6 and 1.5-7.1 kg-CO_2eq/m^3-sludge [26]. Although sewage sludge has long been regarded as waste, it is being reconsidered for use as a biomass resource for energy utilization and as a means to control global warming effects [8, 26]. Anaerobic digestion coupled to power generation processes can be expected to result in excess energy production if high sludge-loading rates are applied [26]. Depending on the recognition of sewage sludge as either waste or a resource, evaluation of treatment processes of WWTPs will change. Strategies for maximization of excess sludge production in WWTPs with low EP and GWP values might be necessary.

5. Conclusions

Treatment performance and greenhouse gas emissions of various biological sewage treatment processes in Japan were evaluated. The conclusions of this research can be summarized as follows.

1) In sewage in Japan, the average BOD concentration was 180 mg/L. All treatment processes in WWTPs can decrease the BOD concentration to < 5 mg/L.

2) The EP value of untreated sewage was estimated as 22 g-PO_4eq/m^3. Nutrient removal processes were regarded as lower than the EP values to < 4 g-PO_4eq/m^3. The EP values of other modified processes not specialized for nutrient removal were as high as that of the CAS process (4.9–7.8 g-PO_4eq/m^3).

3) The GWP value of untreated sewage was estimated as 0.31 kg-CO_2eq/m^3. The CAS process reduces the GWP value to 0.22 kg-CO_2eq/m^3. Other treatment processes showed higher GWP values (0.25–0.68 kg-CO_2eq/m^3) than those of the CAS process.

4) Sludge yields of treatment processes without the PST such as the nitrification/endogenous denitrification, OD, and extended aeration processes were lower (0.011–0.024 m^3/m^3) than those of other processes (0.026–0.033 m^3/m^3).

5) The EP values for the nutrient removal processes showed negative correlation with GWP values. The step-feed nitrification–denitrification process with a low EP and GWP values was the sole exception to this tradeoff.

6) Sludge yields of the nutrient removal processes and modified processes for other purposes respectively showed negative correlations with their GWP values.

Acknowledgement

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