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Citation for the original published paper (version of record):
Ekholm, J., de Blois, M., Persson, F. et al (2023). Case study of aerobic granular sludge and activated sludge—Energy usage, footprint, and
nutrient removal. Water Environment Research, 95(8). http://dx.doi.org/10.1002/wer.10914

Received: 30 March 2023 | Revised: 29 June 2023 | Accepted: 23 July 2023
DOI: 10.1002/wer.10914

CASE STUDY

water ENVIRONMENT RESEARCH

Case study of aerobic granular sludge and activated sludge—Energy usage, footprint, and nutrient removal

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Funding information
Gryaab; H2OLAND; Käppala Association; Svenskt Vatten; Sweden Water Research; Uppsala Vatten och Avfall

Abstract

This study demonstrates a comparison of energy usage, land footprint, and volumetric requirements of municipal wastewater treatment with aerobic granular sludge (AGS) and conventional activated sludge (CAS) at a full-scale wastewater treatment plant characterized by large fluctuations in nutrient loadings and temperature. The concentration of organic matter in the influent to the AGS was increased by means of hydrolysis and bypassing the pre-settler. Both treatment lines produced effluent concentrations below 5 mg BOD_7 L^{-1}, 10 mg TN L^{-1}, and 1 mg TP L^{-1}, by enhanced biological nitrogen- and phosphorus removal. In this case study, the averages of volumetric energy usage over 1 year were 0.22 ± 0.08 and 0.26 ± 0.07 kWh m^{-3} for the AGS and CAS, respectively. A larger difference was observed for the energy usage per reduced population equivalents (P.E.), which was on average 0.19 ± 0.08 kWh P.E.$^{-1}$ for the AGS and 0.30 ± 0.08 kWh P.E.$^{-1}$ for the CAS. However, both processes had the potential for decreased energy usage. Over 1 year, both processes showed similar fluctuations in energy usage, related to variations in loading, temperature, and DO. The AGS had a lower specific area, 0.3 m^2 m^{-3} d^{-1}, compared to 0.6 m^2 m^{-3} d^{-1} of the CAS, and also a lower specific volume, 1.3 m^3 m^{-3} d^{-1} compared to 2.0 m^3 m^{-3} d^{-1}. This study confirms that AGS at full-scale can be compact and still have comparable energy usage as CAS.

Practitioner Points

- Full-scale case study comparison of aerobic granular sludge (AGS) and conventional activated sludge (CAS), operated in parallel.
- AGS had 50 % lower footprint compared to CAS.
- Energy usage was lower in the AGS, but both processes had potential to improve the energy usage efficiency.

Electronic supplementary information (ESI) available.

- Both processes showed low average effluent concentrations.

KEYWORDS

activated sludge, aerobic granular sludge, biological nutrient removal, electricity usage, full-scale wastewater treatment, land footprint, volume requirement

INTRODUCTION

Clean water, sanitation, sustainable cities, and climate action are goals for sustainable development (SDG; (Nations, 2022) that need to be examined and addressed in the water sector. Additionally, the pressures of growing cities, competition for use of urban areas, and higher influent loads are pushing for innovative technologies for treatment with low demands for land footprint and costs (Winkler & van Loosdrecht, 2022). Future (new and retrofitted) wastewater treatment plants (WWTPs) have the potential to exhibit improved environmental performance and contribute to the achievement of the SDG (Seifert et al., 2019). Stricter effluent quality requirements and requirements on positive energy balance are expected in the future, which is why new plants need to produce improved effluent quality while at the same time minimizing energy usage. Technologies based on aerobic granular sludge (AGS), holds some advantages compared to the conventional activated sludge (CAS) with respect to these pressures. The main difference between the two is the morphology of the microbial aggregates. Compared to the flocculent biomass typically produced in a CAS system (Andreadakis, 1993), aerobic granules are larger and more compact and therefore settle more rapidly. Reactors for AGS have a typical design with a single tank volume for all biological reactions as well as settling. Higher biomass concentrations can be applied compared to CAS and separate settling tanks are avoided (Bengtsson et al., 2018; Hamza et al., 2022). With a typical design, an AGS system will normally have a 40%–50% reduced land footprint compared to a CAS system (Bengtsson et al., 2019).

The energy usage of a plant can vary greatly depending on details of the configuration as well as the mechanical equipment. At full-scale, 50% lower energy usage for an AGS system compared to a parallel activated sludge plant was observed (Pronk et al., 2015). However, energy comparison has only been demonstrated for few installations. At Garmerwolde WWTP (NL), the energy comparison was done for 10 months including 3 months of start-up and was hence not covering the fluctuations in operational performance during a whole year (Pronk et al., 2015). Biological nitrogen and phosphorus removal rely on the availability of organic substrates (Henze et al., 2019). The Dutch AGS plant had relatively high concentrations of organic matter in the influent (average concentrations of chemical oxygen demand [COD] 506 mg L^{-1}, biological oxygen demand [BOD] of 224 BOD_5 mg L^{-1}, and BOD/N-ratio of 4.5), compared to the full-scale AGS plant in Sweden (average concentrations of COD 277 mg L^{-1}, BOD_7 110 mg L^{-1} and BOD/N-ratio of 3.2; Ekholm et al., 2022), which influences the process performance.

Compared to the relatively new AGS technology, CAS processes have been widely studied regarding energy usage, biogas production, and environmental impacts (Foladori et al., 2015; Hao et al., 2018; Llácer-Iglesias et al., 2021; Silva & Rosa, 2022; Yang et al., 2010). For example, aeration efficiency has been extensively studied for CAS systems (Baquero-Rodríguez et al., 2018) but scarcely for AGS processes (Strubbe et al., 2023). Similarly, the biogas production process from activated sludge has been broadly studied regarding methods and improvements (Uthirakrishnan et al., 2022), but the studies of waste AGS biogas potential are few (Bernat et al., 2017; Cydzik-Kwiatkowska et al., 2022; Guo et al., 2020; Jahn et al., 2019). Biogas can be produced in anaerobic digestion from the excess sludge, which can positively impact the energy balance (Gude, 2015) and thereby contribute to the sustainability of the WWTP. The biodegradability of waste sludge from AGS has been observed to be lower than that of CAS (Guo et al., 2020), but in another study, the methane production was found to be in the same range for AGS as for CAS (Jahn et al., 2019). However, whether the amount of sludge production is significantly different between AGS and CAS has not been studied. Consumption of chemicals is another factor that will impact the sustainability of the process and in this respect, the extent of enhanced biological phosphorus removal (EBPR) is highly relevant.

In summary, the available knowledge of full-scale AGS operation, land footprint, waste sludge management, and process performance including energy usage is still limited (Bengtsson et al., 2018). Because of the potential positive contribution to the SDG, studies of wastewater treatment technologies are needed, especially at full scale with real variations in environmental conditions and loadings. In this paper, the aim was to increase the knowledge of full-scale operation of AGS for municipal wastewater treatment, by comparing parallel processes based on AGS and CAS, with special attention to energy usage, footprint, and treatment performance. The AGS system comprised sequencing batch reactors whereas the CAS system was based on tanks in series with continuous flow. The case study was performed at the Österröd WWTP (Sweden), characterized by large fluctuations in nutrient loads and temperature (Ekholm et al., 2022). Effluent quality, land footprint, energy usage, and sludge production were compared for the two techniques. The energy usage was divided into different operational categories such as aeration, mixing, and pumping, to analyze the potential for optimizations.

MATERIALS AND METHODS

Description of the WWTP

The studied WWTP (Österröd WWTP) is located in Strömstad, Sweden (58°55′59.1″N 11°11′48.8″E), designed for 30,000 population equivalents (P.E.) during the summer (high season with a large increase in the organic and nutrient load due to tourism) and 15,000 P.E. from autumn to spring.

The wastewater line consists of inlet screens (6 mm), an aerated fat- and sand trap, a flocculation tank, primary settlers, biological treatment, a flocculation tank for the effluents of the parallel AGS and CAS (dosage of poly-aluminum chloride as precipitation chemical), and a final settler (Figure 1). The AGS line is composed of two sequencing batch reactors (R1 and R2), and influent and effluent buffers. One of the pre-settling tanks is used to supply only the AGS, in which also in-line hydrolysis/fermentation is applied. One of the pre-settling tanks supplies both the AGS and the CAS, and one is solely used at peak flow conditions. The pre-settlers removed 40%–50% of the influent COD. Part of the flow to the AGS was bypassed the pre-settlers. Data from the two treatment lines were analyzed from October 2020 to the end of September 2021, which coincided with a process guarantee period stating maximum effluent concentrations from the AGS of 8 mg L^{-1} of BOD_7, 10 mg L^{-1} of total nitrogen (TN), and 1 mg L^{-1} of total phosphorus (TP) as yearly average and average May–August.

Rejection water from the influent screens, primary sludge, excess sludge from the AGS, the CAS, chemical sludge from the final settler and floating sludge are, after storage, mixed and treated in two aerated or mixed sludge storage tanks for stabilization or hydrolysis. External sludge is treated by screens and a sand-trap whereafter the sludge, after storage, is aerated or hydrolyzed together with the excess sludge. The mixed, treated sludge is then led to a mechanical thickener and a screw press after polymer addition. Reject water from dewatering (concentrations given in Table S1) is led to the primary settlers, activated sludge reactor, or to the flocculation before the final settler. The dewatered sludge is transported away and spread on agricultural land.

AGS and CAS process properties

The biological treatment consists of parallel lines for AGS (built 2017–2018) and CAS (renovated 2018–2019). The volumes and footprints of the tanks for the AGS and CAS lines are given in Table 3. The two AGS reactors of 758 m^3 each and a depth of 7 m are equipped with sensors for temperature, redox potential, dissolved oxygen (DO), pH, suspended solids, and nitrate concentrations, and analysers for measurements of ammonium and phosphate concentrations every 10–14 min (Endress + Hauser). Each of the two aeration systems contains 360 disc diffusers with 20% degree of coverage (Sulzer PIK S D88,9) and they are supplied by compressed air from four rotary lobe blowers (Kaeser DB166C, 30 kW).

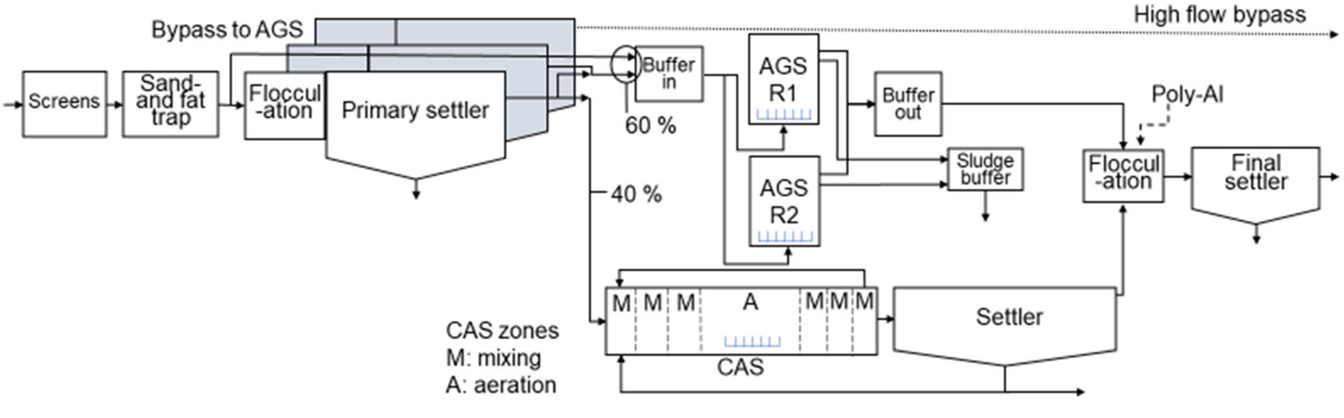

FIGURE 1 Simplified process scheme of the Österröd wastewater treatment plant (WWTP).

The DO setpoint was normally 2 mg L^{-1}, which was increased to 3 mg L^{-1} at short cycle times.

The CAS reactor is 3.6 m deep, has a total volume of 1300 m^3 and is designed for both pre-and post-denitrification. The aerated volume is 520 m^3 (Zone 4), the pre-denitrification volume is 470 m^3 (Zone 1–3) and the post-denitrification volume (Zone 5–7) is in total 310 m^3. In the CAS reactors, DO, ammonium, suspended solids, and nitrate concentrations are monitored on-line. The number of zones, which were aerated, was adapted automatically depending on the ammonium content in the effluent, but normally only Zone 4 was aerated. The CAS line has a secondary settler with surface area of 380 m^2 and a volume of 1290 m^3, equipped with a sensor measuring suspended solids. The aeration system contains 339 disc diffusers (170 in Zone 4, with 20% degree of coverage; IFU 520 ABK/IFU diffuser 02-GIGANT), and the air is supplied from three rotary lobe blowers (Kaeser DB166C, 18.5 kW).

Sampling of water and sludge

Influent and effluent flow proportional water samples were collected regularly from the CAS and AGS lines except from the CAS in May 2021. In the AGS, sampling of sludge was done at the depth of 1.5 and 5 m (mixed into one collected sample), and in the CAS at the depth of 1 m. A Ruttner sampler was used for sludge sampling during aerated conditions.

Analyses of wastewater and sludge

Analyses of BOD_7, COD, ammonium-nitrogen, nitrate-nitrogen, nitrite-nitrogen, total Kjeldahl nitrogen (TKN), TN, TP, and phosphate-phosphorus were performed on influent and effluent samples according to standard methods (APHA 1992). The soluble fraction of the COD was determined after filtration through 0.45 μm pore size filters.

The sludge properties were analyzed by standard methods for total suspended solids (TSS), volatile suspended solids (VSS), total solids (TS), volatile solids (VS), and sludge volume index (SVI) after 10 and 30 min (APHA 1992). Diluted SVI was analyzed for CAS (40% sludge, 60% effluent). The size distribution of the sludge and granules was determined by sieving 1 L of sludge through a series of sieves with pore sizes of 2, 1.4, 0.6, 0.4, and 0.2 mm. The washed sample remaining on each sieve was dried at 105°C. The fraction <0.2 mm was calculated by subtracting the sum of the sieved samples

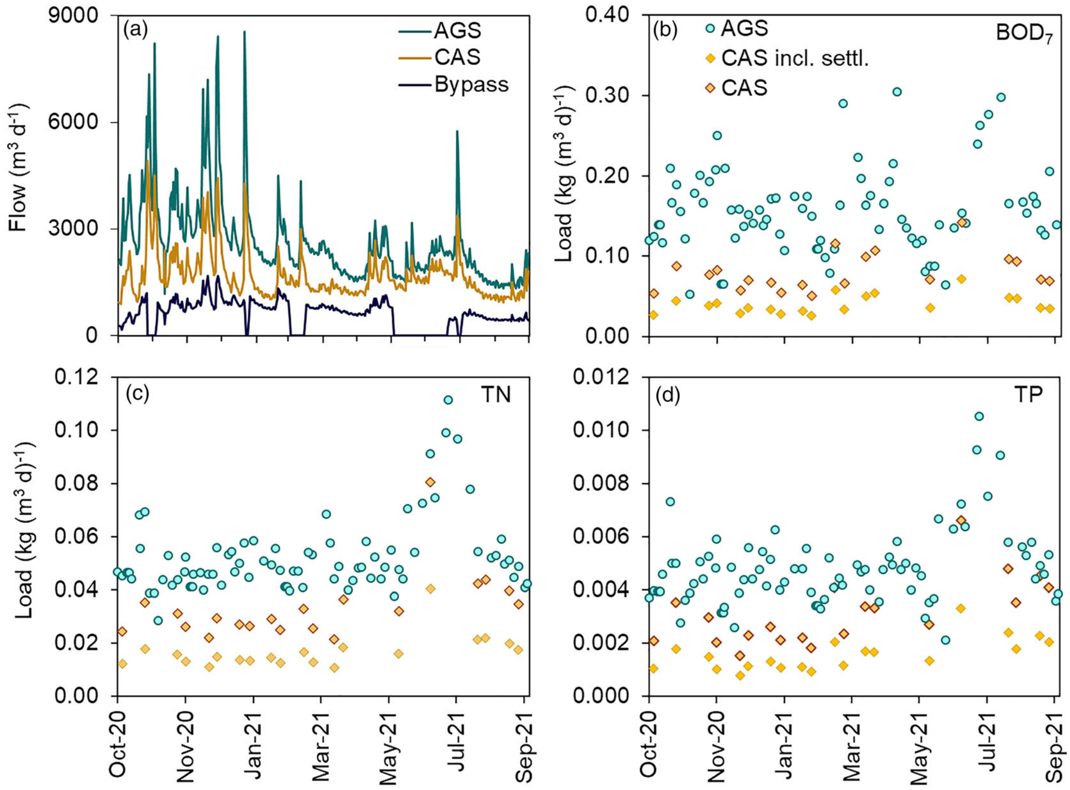

FIGURE 2 Total flow and load to conventional activated sludge (CAS) and aerobic granular sludge (AGS) (a) total flow and bypassed flow (to the AGS); (b) load of BOD_7; (c) load of total nitrogen (TN); and (d) load of total phosphorus (TP).

from the TSS. The sludge samples were examined monthly by light microscopy (Olympus BX53) with micrographs taken by a digital camera (Olympus DP11).

Influent wastewater characteristics

The plant receives mainly domestic wastewater and the partly combined sewer system as well as storm water intrusion resulted in several-fold increased inflow rates at rain and snow-melt events (Figure 2A). The influent wastewaters entering the two biological treatment lines were different (Figure S1, Tables S2, S3, and S4) since the wastewater to the AGS had been treated with in-line hydrolysis/fermentation and a part of the influent was bypassed the pre-settlers (on average 24% ± 14% of the flow to the AGS) to increase the load of organic matter. The influent to the CAS was pre-settled, with no hydrolysis/fermentation. The influent wastewater concentrations of TN and TP were generally similar between the lines, except for one occasion in July 2021 when the AGS received a peak concentration of around 75 mg TN L^{-1}. The COD and BOD_7 concentrations were higher in the AGS influent, as were the BOD_7/TN- and BOD_7/TP-ratios, due to the bypassed flow and the hydrolysis in the pre-settling feeding the AGS (Figure S1, Table S2).

Reactor operation and loads

The flow was typically divided according to the design, that is, 60% (±4%) to the AGS and 40% to the CAS (Figure 2A). The average flow to the AGS was 2636 ± 1180 m^3 d^{-1} and to the CAS 1683 ± 656 m^3 d^{-1}. The operational parameters of the AGS and CAS reactors are summarized in Table 1. The loads of BOD_7, TN, and TP were typically higher to the AGS compared to the CAS, and followed seasonal variations (Figure 2B,C,D).

TABLE 1 Operational parameters of the AGS and CAS reactors, average ± standard deviation from October 2020 to (and including) September 2021.

Parameter	Unit	AGS R1	AGS R2	CAS
Solids retention time	d	26–39[a]	24–41[a]	25–48[a]
Sludge concentration	g TSS L^{-1}	8.6 ± 0.8	8.8 ± 0.6	3.3 ± 0.6
Biomass ratio	VSS/TSS	0.86 ± 0.02	0.86 ± 0.02	0.86 ± 0.04
SVI_{10}	mL/g TSS	49 ± 5	48 ± 4	394 ± 139
SVI_{30}	mL/g TSS	47 ± 6	46 ± 4	262 ± 120
Dissolved oxygen	mg/L	2.02 ± 0.4	2.02 ± 0.4	1.99 ± 0.7
Parameter	**Unit**	**AGS (R1 + R2)**		**CAS**
Average temperature	°C (min−max)	13.2 ± 3.9 (6.6–20.6)		-
Influent pH	-	8.2 ± 1.0		-
BOD_7/N-ratio	-	3.0 ± 0.8		2.6 ± 0.5
BOD_7/P-ratio	-	33 ± 9		25.8 ± 3.5
F/M-ratio[b]	kg BOD_7 $(kg\ TSS\ d)^{-1}$	0.017 ± 0.006		0.024 ± 0.007
P.E. load[c]	P.E.	3300		1500
P.E. reduced	P.E.	3100		1400
Return sludge flow	m^3 h^{-1}	-		108 ± 54
Cycle time	h (min−max)	7.7 ± 2.1 (2.2–11.6)		-
Exchange ratio	-	0.46 ± 0.05		-
Feed velocity	m h^{-1}	3.44 ± 0.03		-
Hydraulic retention time	h^{-1} (min−max)	16.6 ± 5.8		41 ± 12 (21 ± 6)[d]

Abbreviations: AGS, aerobic granular sludge; BOD, biological oxygen demand; CAS, conventional activated sludge; P.E., population equivalent; SVI, sludge volume index; TSS, total suspended solid.

[a] Calculated from averages. The range was calculated from min- and max sludge concentrations.

[b] Based on kg BOD_7 divided by the total amount of sludge in the reactor. The average sludge concentration of R1 and R2 was used for the AGS.

[c] Based on average BOD_7-load and 70 g BOD_7 per person and day.

[d] Values in parenthesis include the CAS and its secondary settler.

Operation of the AGS

The AGS reactors were operated with cycles consisting of anaerobic feeding and simultaneous decanting (50–60 min), sludge discharge, pulse aeration for pre-denitrification (5–30 min, based on nitrate concentration), aeration (30–120 min, based on ammonium concentration), pulse aeration for post-denitrification (based on time left of the cycle), stripping (10–20 min), and settling (20 min) with sludge discharge (only at long cycle times). A control system (Aquasuite® Nereda controller) was used to determine the total cycle time based on the influent flow rate. The up-flow (feeding) velocity was typically 3.5 m h^{-1}, and the exchange ratio was 35%–50% depending on the influent flow rate (Table 1). The DO concentration was close to 2 mg L^{-1} on average (Table 1). A mixed sludge discharge (during aeration) was done typically one to three times per month to control the sludge concentration and size distribution.

Operation of the CAS

The wastewater was fed to the CAS by gravity to Zone 1, into which also the return sludge was pumped. Zones 1–3 and 5–7 were mixed and under anoxic conditions for denitrification. Zone 4 was aerated (average DO 1.7 mg L^{-1}) for nitrification and organic matter removal. Nitrate recirculation was applied from Zone 4 to 1 until February 19, 2021 and thereafter, to Zone 2. Operation for post-denitrification was limited, as feeding of influent wastewater to Zone 5–7 to increase the available organic matter for post-denitrification was not in operation. The temperature was not logged in the CAS but assumed to be the same as in the AGS.

Data collection

On average, 55% of the flow from the pre-settlers was pumped to the AGS, and the remaining 45% was entering the CAS. Hence, the volume and footprint of the pre-settlers were calculated accordingly for each process. For the AGS, 60% of the volume and footprint for the shared flocculation and final settler was considered, and 40% for the CAS (Table 3).

Data on energy usage (kWh day^{-1}) was collected from the SCADA system. The period from October 1, 2020 to September 30, 2021 was considered. The sludge treatment process (dewatering and thickening) was not taken into account, as it received sludge from both the CAS and the AGS, which are mixed in the sludge storage. Determining the contribution of each sludge (CAS, AGS) to the total energy usage was therefore not possible. Additional components that were not directly linked to the processes are the heating system, dehumidifier, ventilation, analyzers, hydrolysis pump, and compressor, and these were also excluded from the comparison. The energy usage was calculated based on removed P.E. and daily average flow rate. Energy usage per reduced load of P.E. was calculated for days with BOD_7 concentrations available and was based on a BOD_7-load of 70 g BOD_7 per P.E. and day.

The components included in the energy comparison were blowers, mixers, pumps, and sludge scrapers (Table S5), and were grouped into the categories of aeration, mixing, sludge pumping, feeding (AGS), and sedimentation (CAS). In the sedimentation category for the CAS, a sludge pump and sludge scraper were included. In the AGS line, a mixer was used in the incoming buffer to prevent sedimentation whereas mixing in the reactor was done with pulse aeration.

The chemical demand was estimated based on the yearly consumption of precipitation chemical, flow rate to the AGS and CAS, and the load of TP from each line entering the second flocculation tank.

Statistical methods

The effluent values from the CAS and AGS were compared statistically as paired samples on the same day or the closest day possible (for a few samples), and the two-tail *p*-values were calculated. The daily energy usage was compared as paired samples. The energy usage and pollutants removals (BOD_7, TP and TN), as well as temperature, DO concentration, and flow were analyzed with linear regression to assess for potential correlations.

RESULTS

Process performance and operation

The conditions at the WWTP were characterized by high variations in flow rates (1936–12,474 m^3 day^{-1}) and temperature (6.6–19.4°C). The flow ratio was kept constant around 60%/40% to the AGS/CAS and thus, both lines experienced similar dynamics in flow rates. The influent loads of nitrogen and phosphorus followed a seasonal trend with increased loads in the summer months. The volumetric loading rates of BOD_7, TN, and TP to the AGS were significantly higher than to the CAS (Figure 2B,C and D), but the biomass specific loadings were overall similar (Figure S2). Organic matter removal, assimilation, nitrification, denitrification, and EBPR produced low effluent concentrations in both processes (Table 2). The effluent concentrations of TN, PO_4^{3-}—P, and TP from

TABLE 2 Effluent wastewater concentrations (mg L^{-1}) from the AGS- and CAS reactors, average ± standard deviation from October 2020 to the end of September 2021. *n* is the number of samples. Significant difference is considered for *p*-values <0.05.

Parameter	CAS	AGS	Significant differences between CAS and AGS	Plant
BOD_7	3.2 ± 0.6	4.8 ± 1	Yes	3 ± 0
BOD_7 soluble	3.0 ± 0.0	5.0 ± 1.8	No	[a]
COD	34 ± 4	42 ± 6	Yes	30 ± 5
COD soluble	31 ± 3	34 ± 5	Yes	[a]
TN	9.7 ± 3	8.7 ± 4	No	8.3 ± 2
NH_4^+-N	0.33 ± 0.3	1.2 ± 1.6	Yes	0.46 ± 0.6
NO_3^--N	8.7 ± 3	6.4 ± 3	Yes	7.1 ± 2
TP	0.76 ± 0.74	0.65 ± 0.45	No	0.21 ± 0.26
PO_4^{-3}-P soluble	0.59 ± 0.68	0.45 ± 0.41	No	[a]
SS	6.6 ± 4	11 ± 3	Yes	7.8 ± 4
n	23	23		58

Abbreviations: AGS, aerobic granular sludge; BOD, biological oxygen demand; CAS, conventional activated sludge; COD, chemical oxygen demand; TN, total nitrogen; TP, total phosphorus.

[a] Not measured.

the AGS and CAS were similar ($p > 0.05$), the concentrations of NO_3^--N were higher in the effluent from the CAS, and the concentrations of BOD_7, COD, NH_4^+-N, and TSS were higher from the AGS ($p < 0.05$). The higher effluent concentration of NH_4^+-N depended on the set-point of NH_4^+-N (1–3.5 mg L^{-1}) at which the aeration was terminated. The final effluent concentrations from the plant are presented in Table 2.

The specific removal rates of organic matter, nitrogen, and phosphorus were similar between the CAS and the AGS ($p > 0.05$), whereas the volumetric removal rates were generally higher for the AGS throughout the study ($p < 0.05$) (Figure S3). The extent of EBPR occurring was estimated by assuming assimilation of 1.5% phosphorus of TSS based on a sludge production of 0.45 kg TSS (kg influent COD)$^{-1}$. The difference between TP removal and phosphorus assimilation was assumed to be due to EBPR. In the AGS, EBPR accounted for an average of 29% of the TP removal, and in the CAS, EBPR accounted for 18%. Denitrification and EBPR were found to be periodically limited by the available organic matter. Bypass of the pre-settler feeding the AGS was implemented as means to provide additional organic matter and facilitate more extensive nitrogen and phosphorus removal. Two periods with elevated effluent concentrations of nitrate and phosphate from the AGS were observed (winter of 2020/2021 and in summer 2021). The reasons behind this were interpreted to be low temperatures, leading to low hydrolysis rates in the pre-settler and in the sewer system, high flows resulting in short cycle times (winter) and temporary lack of bypass flow (summer) leading to lack in organic substrate. The average removal of TN was 73% ± 13% in the AGS and 65% ± 24% in the CAS.

Energy usage

The relative electricity usage per P.E. for the whole WWTP was 210 kWh P.E.$^{-1}$ in 2020 and 160 kWh P.E.$^{-1}$ in 2021. This was higher than before the re-construction (110–160 kWh P.E.$^{-1}$ over the years 2012–2017), since the plant was rebuilt with overcapacity, leading to higher fixed energy usage by machines, other equipment, and heating. The total energy usage of directly associated equipment during the year (October 1, 2020 to September 30, 2021) was 147 MWh for the CAS and 191 MWh for the AGS. The AGS treated 60% of the flow and used 57% of the total average energy usage for both processes. The energy usage per m^3 of treated wastewater was lower for the AGS ($p > 0.05$), on average 15% lower than for the CAS, 0.22 ± 0.08 kWh m^{-3} for the AGS compared to 0.26 ± 0.07 kWh m^{-3} for the CAS. Furthermore, a lower ($p < 0.05$) energy usage per reduced P.E. was found for the AGS, 0.19 ± 0.08 kWh P.E.$^{-1}$ compared to 0.30 ± 0.08 kWh P.E.$^{-1}$ for the CAS. The monthly averages were often higher for the CAS than for the AGS relative to the flow and the load (Figure 3A,B, and C). The volumetric energy usage for aeration in the CAS and AGS were correlated ($p < 0.05$) (Figure 3D).

Energy usage for different categories

Most of the energy usage in the AGS was for aeration, including pulse aeration for mixing (71% on average), followed by feeding (16%), mixing (12%), and sludge pumping (1%). For the CAS, mixing and aeration accounted for 51% and 42%, respectively, followed by sludge pumping

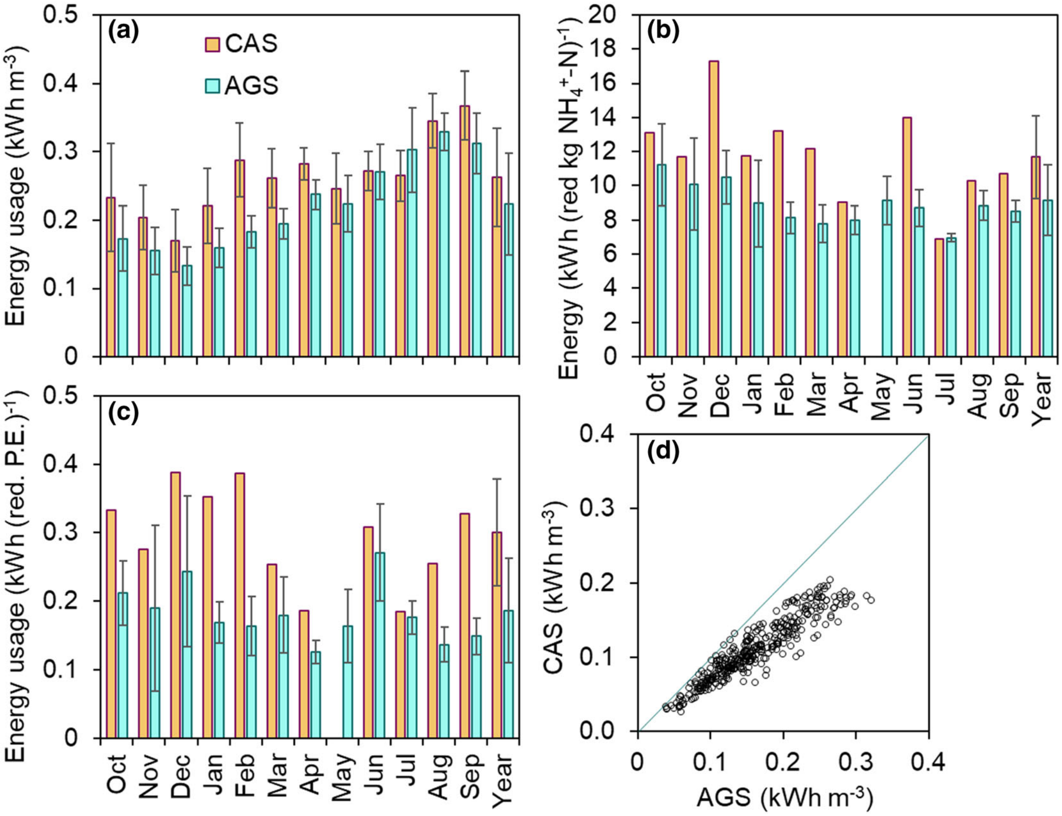

FIGURE 3 Energy usage for conventional activated sludge (CAS) and aerobic granular sludge (AGS) compared in terms of (a) kWh m^{-3} treated wastewater, (b) kWh per reduced kg NH_4^+-N, (c) kWh P.E.$^{-1}$ based on reduced load of BOD_7 (data on P.E. [BOD_7] is missing for the CAS in May), and (d) volumetric aeration energy usage for the CAS versus the AGS.

(6%) and sedimentation (1%). The influent was entering the CAS by gravity; hence, no energy usage was needed for feeding. The energy usage of aeration and mixing was dynamic over the year for both processes (Figure 4). The highest volumetric energy usage was in August and September, which was due to higher concentrations of BOD_7, COD, TN, and TP during the summer months (Figure S1, Table S3, and S4). The average energy usages per reduced P.E. were higher for the months with lower loads of P.E., December to February for the CAS, and June for the AGS (Figure 4C and D).

Energy usage related to aeration

The aeration energy per m^3 varied with the DO concentration and seasonally in the AGS and the CAS (Figure S4). The energy usage in both processes was increasing with the temperature (Figure S5). This was both a direct effect of the temperature (lower saturation concentration of oxygen) and an effect of the higher pollutant concentrations during summer. No correlation was observed between the aeration energy usage in the AGS and the bypassed flow (Figure S6). The removal of BOD_7, NH_4^+-N, and TP were compared with the energy usage for aeration in the AGS and CAS, respectively, as removed kg d^{-1} in relation to kWh d^{-1} and as removed mg L^{-1} in relation to kWh m^{-3}. As could be expected, the aeration energy usage increased with the removal of pollutants (Figures S7–S10). The removal of NH_4^+-N had the best fit in both systems, which is in line with the higher stoichiometric oxygen demand for nitrification compared with organic matter removal (Henze et al., 2019). Thus, the high energy usage at high removal rates was likely mainly linked to more extensive nitrification.

Land footprint and volume

The specific land footprint, that is, area per treated flow of wastewater (m^2 m^{-3} d^{-1}) was about 50% lower for the AGS than for the CAS (total area) and 70% lower when excluding the area of the flocculation, pre-settler, and final sedimentation. Although the volumes of the two processes were similar, the specific volume (m^3 m^{-3} d^{-1}) was 36% lower for the AGS (Table 3). The total area footprint including all the components related to each

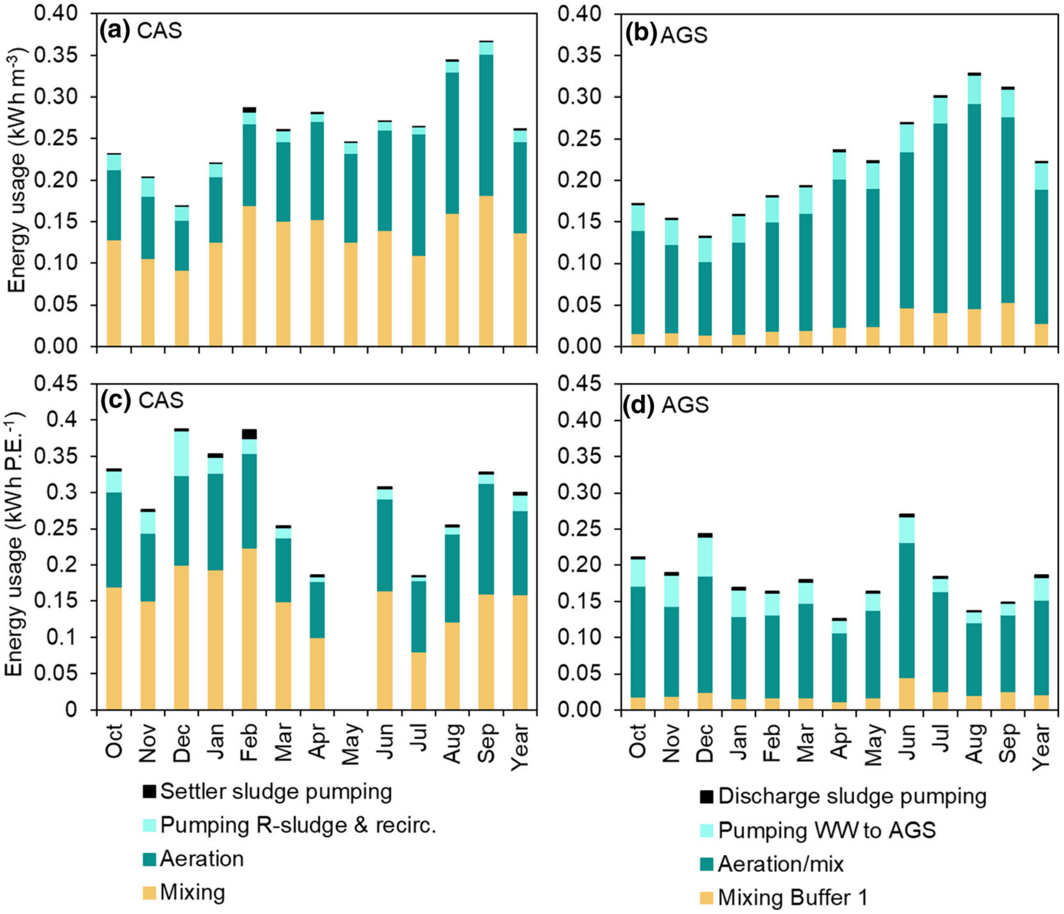

FIGURE 4 Monthly averages of volumetric energy usage for (a) the conventional activated sludge (CAS) and (b) the aerobic granular sludge (AGS), and energy usage per reduced population equivalent (P.E.) for (c) the CAS, and (d) the AGS, divided into the energy categories of pumping, mixing, and aeration.

biological treatment line was 24% lower for the AGS line than for the CAS, and 50% lower for the AGS if the flocculation, pre-settler, and final sedimentation were excluded (Table 3). The higher sludge concentration allowing smaller volumes, absence of a settler and deeper reactors for the AGS process contributed to the lower footprint. The CAS was refurbished in an existing volume and could theoretically have been deeper. If the CAS would have been equally deep as the AGS reactors (7 m), the land footprint per treated m^3 of wastewater (total area) for the AGS would have been 40% instead of 50% lower than for the CAS.

Sludge characteristics and sludge production

The sludge concentration was similar in the two AGS reactors (R1 8.6 ± 0.6 and R2 8.8 ± 0.6 mg L^{-1}) and lower in the CAS (3.3 ± 0.6 mg L^{-1}) (Table 1). The SVI$_{30}$ was very stable and low in the AGS, as opposed to the CAS, which had varying and high values of diluted SVI (Table 1). The organic fraction of the biomass (VSS/TSS) was 0.86 in all reactors. The granular size distributions in R1 and R2 had a majority of granules >2 mm. In R1, 80% ± 7% of the biomass was >2 mm and in R2, it was 86% ± 7%. The total mass of sludge per treated m^3 of treated wastewater was 47% ± 16% higher in the AGS compared to the CAS. The two processes had similar average solids retention time (SRT) (Table 1); however, the SRT in AGS varies greatly with aggregate sizes (Ali et al., 2019).

The sludge production at Österröd WWTP was assessed. The total sludge production of the whole WWTP ranged from 270 to 550 tons per year for the years 2011–2021. The specific sludge production was 0.12 ± 0.03 kg TS P.E.$^{-1}$ before the implementation of the AGS. In 2020–2021, with AGS in operation, it was similar and on average 0.13 kg TS P.E.$^{-1}$.

TABLE 3 Footprint and volume for AGS and CAS line.

AGS line	Footprint (m^2)	Volume (m^3)	Specific footprint ($m^2\ m^{-3}\ d^{-1}$)	Specific volume ($m^3\ m^{-3}\ d^{-1}$)
Flocculation	20	70	0.0076	0.027
Pre-settler (together with CAS)[a]	110	440	0.042	0.17
Buffer 1	70	340	0.027	0.13
AGS reactors	220	1510	0.083	0.57
Buffer 2	70	450	0.027	0.17
Sludge buffer	10	30	0.038	0.011
Flocculation (together with CAS)[b]	20	80	0.0076	0.019
Final settler (together with CAS)[b]	270	550	0.10	0.21
Total	790	3440	0.30	1.3
Total without floccul. pre-settl. and final settl.	370	2330	0.14	0.88
CAS line				
Flocculation	20	70	0.012	0.042
Pre-settler (together with AGS)[a]	90	360	0.054	0.21
Activated sludge reactor	360	1300	0.21	0.77
Settler for activated sludge	380	1290	0.23	0.77
Flocculation (together with AGS)[b]	10	80	0.0059	0.018
Final settler (together with AGS)[b]	180	360	0.11	0.21
Total	1040	3410	0.62	2.0
Total without floccul. pre-settl. and final settl.	740	2590	0.44	1.5

Abbreviations: AGS, aerobic granular sludge; CAS, conventional activated sludge.

[a] Divided as 55%/45% for the AGS and CAS, respectively.

[b] Divided as 60%/40% for the AGS and CAS, respectively.

TABLE 4 Chemical consumption and energy usage per treated flow of wastewater for AGS and CAS.

	Chemical consumption (kg m^{-3})	Energy usage (kWh m^{-3})
AGS	0.0489	0.22
CAS	0.106	0.26

Abbreviations: AGS, aerobic granular sludge; CAS, conventional activated sludge.

Chemical and maintenance demand

The CAS process was associated with a higher demand of precipitation chemicals, due to the higher load of phosphorus in the effluent (Table 4). However, with the potentially improved operation of EBPR in the CAS, the consumption of precipitation chemicals was expected to be similar for the CAS and the AGS. Compared to the CAS, the AGS had more instrumentation with related costs for reagents and maintenance. However, it can, in general, be assumed that a similar amount of instrumentation is needed in AGS and CAS if an efficient process control for nitrogen and phosphorus removal is aimed for. The maintenance costs of the aeration system and other machinery was estimated to be similar for the two processes.

DISCUSSION

In this study, full-scale parallel CAS and AGS processes were compared along five critical indicators: effluent quality and removal performances, energy usage, land footprint, and volumetric need.

Differences in operation were substantial but led to comparable effluent quality

Despite fundamental differences (batch-wise vs. continuous operation, etc.), the processes delivered similar effluent concentrations of TP and TN (Table 2). The effluent concentrations of BOD_7, COD, and TSS were slightly higher from the AGS compared to the CAS. This was likely related to the higher suspended matter loading during the settling phases of the AGS reactors compared to the continuously operated secondary settler of the CAS. However, the effluent concentrations were still low in comparison with the discharge targets (according to the process guarantee: 8 mg BOD_7 L^{-1}, 70 mg COD L^{-1}). The anoxic volumes of the CAS were not optimally used for post-denitrification, as influent wastewater was not pumped to the zones after the aeration due to technical problems. This meant that the post-denitrification in Zones 5–7 could only utilize internally stored carbon as an electron donor. Nevertheless, the effluent concentration of TN from the CAS was under the discharge limit.

In the AGS, the partial bypass of the pre-settler was used as a measure to increase the BOD_7/N- and BOD_7/P-ratio. The bypassed flow led to significantly higher incoming concentrations of organic matter, TN, TP, BOD_7/N-, and BOD_7/P-ratio to the AGS than to the CAS ($p < 0.05$, Table S2). It was found that the bypass improved the denitrification and EBPR performance in the AGS. For instance, in a period in July 2021 when the bypass was turned off, the effluent concentrations of nitrate-nitrogen and phosphate increased, suggesting that the organic substrate concentration in the pre-settled wastewater was too low to support both denitrification and EBPR. The CAS process reached similarly low TN and TP levels as the AGS (Table 2), without bypass of the pre-settler. High flow rates and low temperatures were other factors that led to increased effluent levels of nitrate and phosphate from the AGS. The high flows led to short cycle times, and as nitrification (main aeration) was prioritized, the time left for denitrification (pulse-aeration) was restricted. This is in line with previous research that suggested that simultaneous nitrification–denitrification (SND) in AGS was limited by anoxic zones in the granules and availability of organic substrate (Layer et al., 2020).

The nitrous oxide emissions were not measured but previous results from a full-scale AGS plant showed an average (7 months) emission factor of 0.33% of TN in the influent, which in the same range or lower compared to activated sludge systems (van Dijk et al., 2021).

Phosphorus removal was similar in the CAS and the AGS

The effluent concentrations of TP from the CAS and the AGS were similar (Table 2), which facilitates a comparison. The CAS was not under ideal operation for EBPR given that the return sludge, carrying nitrate, was pumped to Zone 1. Furthermore, the recycle flow of nitrate was pumped to the same zone as the return sludge during the first 4 months of the study. This operational mode may have created an anoxic environment for denitrification. To facilitate uptake of organic matter and phosphorus release by EBPR, the CAS would have to be operated under anaerobic conditions in the tank volume where the influent wastewater enters the reactor. However, the TP removal in the CAS indicated that EBPR was occurring alongside phosphorus assimilation. In the AGS, elevated effluent phosphorus concentrations (1–1.8 mg TP L^{-1}) were related to the colder months and when the bypass flow was turned off, and was likely caused by limited availability of organic substrate. The dense biomass structure in the AGS creates a diffusion resistance, which limits the mass transport into the granular core (de Kreuk et al., 2010). For the organic substrate to be available for the AGS, it needs to be in the form of diffusible compounds. The formation of diffusible compounds by hydrolysis of particulate organic substrate in the sewers, pre-settler and during the anaerobic feeding was likely decreased at lower temperatures (Jönsson et al., 1996). Nevertheless, the AGS delivered a yearly average TP concentration of 0.65 ± 0.45 mg L^{-1}. A phosphorus concentration of 0.65 mg L^{-1} is however above typical effluent permits in Sweden, which is why the combined effluent phosphorus from the CAS and the AGS was precipitated and directed to a final settler. The effluent from the final settler was on average 0.21 ± 0.22 mg L^{-1}.

Energy usage and compactness

The monthly averages of both volumetric energy usage and energy usage per reduced NH_4^+-N and reduced P.E. varied and were often higher for the CAS than the AGS (Figure 3B and C). The variation over the year likely depended on the fluctuations of the flow rate, influent concentrations, removal performance, temperature, and DO concentration (Figures S4–S10). These factors varied simultaneously and thus, the contribution of a single factor is difficult to assess. The CAS and AGS volumetric energy usage per day for aeration were correlated ($p < 0.05$; Figure 3D), suggesting that the main factors that influenced the aeration energy demand affected both systems in similar ways.

The higher volumetric aeration energy usage for the AGS was caused by several factors (Figure 4A and B). A part of the aeration energy (20%) was used for mixing purposes during pulse-aeration and nitrogen gas stripping and was hence not directly related to the main aeration phases. Furthermore, the higher depth of the AGS reactors contributed to a higher energy usage since the specific energy usage per volume of air increases sharply with the counterpressure for rotary lobe type of blowers (AtlasCopco, 2015; van Leuven et al., 2010) as further detailed below. A higher depth increases the oxygen transfer efficiency, which can be offset by the increased pressure that leads to higher energy usage per volume of air (Baquero-Rodríguez et al., 2018). In a theoretical comparison of AGS and CAS with EBPR where the same water depth was considered (5 m), the aeration energy usage was estimated to be similar in the two systems (Bengtsson et al., 2019). Additionally, the higher influent concentrations of organic matter and nitrogen were likely resulting in a small contribution to a higher demand of air. The average DO concentrations in the two systems were similar, on average 2 mg L^{-1} (Table 1), thus not a reason for the higher aeration energy usage in the AGS. The sludge concentration in the AGS was slightly higher than the characteristic 8 g L^{-1}, whereas in the CAS, the sludge concentration was lower than the typical design value of 4 g L^{-1}. Previous studies have demonstrated that sludge concentration can adversely affect oxygen transfer efficiency (Krampe & Krauth, 2003); however, a higher SRT can have a positive effect (Rosso et al., 2008). Thus, the combined influence of sludge concentration and SRT on the aeration energy usage is difficult to assess.

Interestingly, the aeration energy in the AGS seemed to decrease at higher DO concentrations, which could be explained by the higher DO set-point at short cycle times (higher flows), which coincided with diluted wastewater and hence decreased the oxygen demand per m^3 (Figure S4A). The aeration energy usage in the CAS seemed to vary irrespectively of the DO concentration, with the exception of July (Figure S4B). Higher energy usage was generally observed at higher temperature (Figure S5). Two temperature phenomena affect the oxygenation of water; the mass transport rate of oxygen increase with the temperature, but the oxygen saturation concentration and thus the driving force decrease with the temperature (Bahadori & Vuthaluru, 2010). These phenomena are counteracting, but the temperature dependence of the oxygen saturation concentration will govern the relationship and lead to higher energy demand at a higher temperature. Furthermore, air has a lower density at a higher temperature, leading to less oxygen in the air, and hence increased energy demand for aeration.

The difference between the lines was larger in kWh $P.E.^{-1}$ than in kWh m^{-3} (Figure 3A and C), due to the higher load of organic matter via the bypassed flow to the AGS. As primary sludge is a potential energy source via biogas production (Ghimire et al., 2021), the bypassed flow was theoretically a loss of energy. In presence of anaerobic digestion, the energy balance for the AGS would have been negatively affected by the reduced amount of primary sludge due to the bypassed flow. The sludge production was similar before and after the AGS was implemented, which means that the potential biogas production and/or handling costs were not influenced by the type of process.

The energy usage was 15% and 38% lower for the AGS compared to the CAS, per treated m^3 wastewater and per reduced P.E., respectively. The absence of mixers, return sludge pumping, and recirculation of wastewater for nitrogen removal has previously been found to result in 30% (Bengtsson et al., 2019) and 21% (Cicekalan et al., 2023) lower volumetric energy usage of AGS compared to CAS with EBPR in theoretical studies. Results from a full-scale plant showed 51% higher volumetric energy usage of the CAS (AB-plant), compared to the AGS (Pronk et al., 2015). The full-scale study was performed at Garmerwolde WWTP in the Netherlands, and the energy usage was 0.17 kWh m^{-3} for the AGS and 0.33 kWh m^{-3} for the activated sludge (AB-plant). In comparison, the AGS at Garmerwolde WWTP had 24% less energy usage per m^3 than the AGS at Österröd WWTP. On the contrary, the CAS at Österröd WWTP had 20% lower energy usage than the AB-plant at Garmerwolde WWTP. AGS and well as CAS plants can have energy usage varying on broad spans depending on process configuration, operating and environmental conditions as well as machine equipment.

Both the AGS and the CAS process at Österröd WWTP have the potential to decrease the operational energy usage. The energy usage for the mixing in the CAS reactor tanks could be decreased by a reduction of the rotation speed, usage of one out of two mixers in Zones 1 and 5, and intermittent mixing. Considerably lower values have previously been measured for mixing energy usage, on average 6.8 W m^{-3} for tanks <200 m^3 and 4.8 W m^{-3} for 200–500 m^3 tanks, in full-scale activated sludge processes in Austria (Füreder et al., 2017). With the values of 6.8 W m^{-3} for tanks <200 m^3 and 4.8 W m^{-3} for 200–500 m^3 tanks for mixing energy usage, the volumetric energy usage for the CAS would have been 24% lower than measured and 10% lower than for the AGS. The post-denitrification Zones 6 and 7 had a very small contribution to the nitrogen removal (data not shown) because no pre-settled influent was added due to a technical failure. The mixers in these zones had limited influence on the removal performance, and an energy usage of 40 kWh together, corresponding to 10% of the total energy usage in the CAS. Without this contribution, the energy usage in the AGS would have been 6% lower compared to the CAS.

The same type of blower was used in both the AGS and the CAS, namely rotary lobe blowers. The energy usage with this type of blower increases more with increasing counterpressure (water depth) than for other types of blower such as screw and dynamic blowers (AtlasCopco, 2015; van Leuven et al., 2010). Thus, given the higher water depth in the AGS reactors, screw blowers or dynamic blower for the AGS reactors would have led to significantly lower energy usage for aeration in these reactors. The water depth, and properties and operation of aeration and mixing equipment are important factors governing energy usage. CAS processes may, as well as processes based on AGS, be designed for minimal energy usage by avoiding mixing and internal recirculation. The results from this study confirm that a process based on AGS can be compact (Table 3) without increasing the energy usage (Figure 3). Other compact technologies such as membrane bioreactor (MBR), moving-bed biofilm reactor (MBBR), and integrated fixed film activated sludge (IFAS) have typically remarkably higher energy usage than CAS (Bengtsson et al., 2019).

CONCLUSIONS

The AGS and CAS both produced effluent quality below 5 mg BOD_7 L^{-1}, 10 mg TN L^{-1}, and 1 mg TP L^{-1} on average. The AGS was found to be more compact than the CAS, with a specific footprint of 0.3 m^2 m^{-3} d^{-1}, compared to 0.6 m^2 m^{-3} d^{-1} for the CAS. The volumetric energy usage was 15% lower for the AGS compared to the CAS, and 38% per reduced P.E. (load of BOD_7). Moreover, the AGS treated higher loads of organic matter, which was increased via bypassing the pre-settlers to improve the EBPR and denitrification. The sludge production of the whole plant remained at a similar level after start-up of the AGS as in the previous 10 years with only CAS, which means that the potential biogas production was expected to be similar for the AGS and CAS. The highest energy category for the AGS was aeration (including pulse-aeration for mixing) whereas the highest category for the CAS was mixing, which depends on site-specific reactor design and operation. The aeration blower type was not ideal for the depth of the AGS reactors and thus, the aeration energy usage has potential to be decreased. For the CAS, the mixing energy usage was higher than typical values for mixing (W m^{-3}) and an updated mixing operation strategy would lead to energy savings. In this case study, the process based on AGS at the Österröd WWTP was found to be compact with lower energy usage compared to the CAS. Other factors than the treatment technology (AGS or CAS) were found to be equally important for reducing energy usage, such as the reactor depth, properties, and operation of the aeration system and mixers.

AUTHOR CONTRIBUTIONS

Jennifer Ekholm: Conceptualization; methodology; formal analysis; investigation; writing—original draft; writing—review and editing. **Mark de Blois**: Conceptualization; methodology; formal analysis; investigation; writing—review and editing; supervision; funding acquisition. **Frank Persson**: Conceptualization; methodology; writing—review and editing; supervision; funding acquisition. **David J. I. Gustavsson**: Conceptualization; methodology; writing—review and editing; supervision; project administration; funding acquisition. **Simon Bengtsson**: Supervision; writing—review and editing; funding acquisition. **Tim van Erp**: Investigation; supervision. **Britt-Marie Wilén**: Conceptualization; methodology; writing—review and editing; supervision; funding acquisition.

ACKNOWLEDGMENTS

We acknowledge the staff at Österröd WWTP for their help and support in sampling and data collection.

Royal HaskoningDHV is acknowledged for their advice and knowledge of the operation of AGS Nereda® systems.

This study was funded by the Swedish Water & Wastewater Association (SVU), Sweden Water Research, H2OLAND, and the Swedish municipal WWTP operators Uppsala Vatten och Avfall, Gryaab, and Käppala Association. Members of the project were as well Chalmers University of Technology, the municipality of Strömstad, the municipality of Tanum, VA SYD, TU Delft and Royal HaskoningDHV and the municipality of Västervik.

CONFLICT OF INTEREST STATEMENT

There are no conflicts to declare.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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REFERENCES

Ali, M., Wang, Z., Salam, K. W., Hari, A. R., Pronk, M., van Loosdrecht, M. C. M., & Saikaly, P. E. (2019). Importance of species sorting and immigration on the bacterial assembly of different-sized aggregates in a full-scale aerobic granular sludge plant. *Environmental Science & Technology*, *53*(14), 8291–8301. https://doi.org/10.1021/acs.est.8b07303

Andreadakis, A. D. (1993). Physical and chemical properties of activated sludge floc. *Water Research*, *27*(12), 1707–1714. https://doi.org/10.1016/0043-1354(93)90107-S

AtlasCopco. (2015). *Compressed air manual* (8th ed.). Atlas Copco Airpower NV.

Bahadori, A., & Vuthaluru, H. B. (2010). Simple Arrhenius-type function accurately predicts dissolved oxygen saturation concentrations in aquatic systems. *Process Safety and Environmental Protection*, *88*(5), 335–340. https://doi.org/10.1016/j.psep.2010.05.002

Baquero-Rodríguez, G. A., Lara-Borrero, J. A., Nolasco, D., & Rosso, D. (2018). A critical review of the factors affecting modeling oxygen transfer by fine-pore diffusers in activated sludge. *Water Environment Research*, *90*(5), 431–441. https://doi.org/10.2175/106143017X15131012152988

Bengtsson, S., de Blois, M., Wilén, B.-M., & Gustavsson, D. (2018). Treatment of municipal wastewater with aerobic granular sludge. *Critical Reviews in Environmental Science and Technology*, *48*(2), 119–166. https://doi.org/10.1080/10643389.2018.1439653

Bengtsson, S., de Blois, M., Wilén, B.-M., & Gustavsson, D. (2019). A comparison of aerobic granular sludge with conventional and compact biological treatment technologies. *Environmental Technology*, *40*(21), 2769–2778. https://doi.org/10.1080/09593330.2018.1452985

Bernat, K., Cydzik-Kwiatkowska, A., Wojnowska-Baryła, I., & Karczewska, M. (2017). Physicochemical properties and biogas productivity of aerobic granular sludge and activated sludge. *Biochemical Engineering Journal*, *117*, 43–51. https://doi.org/10.1016/j.bej.2016.11.002

Cicekalan, B., Kosar, S., Cingoz, S., Eyit, N., Ersahin, M. E., & Ozgun, H. (2023). Techno-economic and environmental assessment of different municipal wastewater treatment systems. *Journal of Water Process Engineering*, *53*, 103822. https://doi.org/10.1016/j.jwpe.2023.103822

Cydzik-Kwiatkowska, A., Bernat, K., Zielińska, M., Gusiatin, M. Z., Wojnowska-Baryła, I., & Kulikowska, D. (2022). Valorization of full-scale waste aerobic granular sludge for biogas production and the characteristics of the digestate. *Chemosphere*, *303*, 135167. https://doi.org/10.1016/j.chemosphere.2022.135167

de Kreuk, M. K., Kishida, N., Tsuneda, S., & van Loosdrecht, M. C. M. (2010). Behavior of polymeric substrates in an aerobic granular sludge system. *Water Research*, *44*, 5929–5938. https://doi.org/10.1016/j.watres.2010.07.033

Ekholm, J., Persson, F., de Blois, M., Modin, O., Pronk, M., van Loosdrecht, M. C. M., Suarez, C., Gustavsson, D. J. I., & Wilén, B.-M. (2022). Full-scale aerobic granular sludge for municipal wastewater treatment—Granule formation, microbial succession, and process performance. *Environmental Science: Water Research & Technology*, *8*(12), 3138–3154. https://doi.org/10.1039/D2EW00653G

Foladori, P., Vaccari, M., & Vitali, F. (2015). Energy audit in small wastewater treatment plants: Methodology, energy consumption indicators, and lessons learned. *Water Science and Technology*, *72*(6), 1007–1015. https://doi.org/10.2166/wst.2015.306

Füreder, K., Svardal, K., Frey, W., Kroiss, H., & Krampe, J. (2017). Energy consumption of agitators in activated sludge tanks—Actual state and optimization potential. *Water Science and Technology*, *77*(3), 800–808. https://doi.org/10.2166/wst.2017.596

Ghimire, U., Sarpong, G., & Gude, V. G. (2021). Transitioning wastewater treatment plants toward circular economy and energy sustainability. *ACS Omega*, *6*(18), 11794–11803. https://doi.org/10.1021/acsomega.0c05827

Gude, V. G. (2015). Energy and water autarky of wastewater treatment and power generation systems. *Renewable and Sustainable Energy Reviews*, *45*, 52–68. https://doi.org/10.1016/j.rser.2015.01.055

Guo, H., van Lier, J. B., & de Kreuk, M. (2020). Digestibility of waste aerobic granular sludge from a full-scale municipal wastewater treatment system. *Water Research*, *173*, 115617. https://doi.org/10.1016/j.watres.2020.115617

Hamza, R., Rabii, A., Ezzahraoui, F.-z., Morgan, G., & Iorhemen, O. T. (2022). A review of the state of development of aerobic granular sludge technology over the last 20 years: Full-scale applications and resource recovery. *Case Studies in Chemical and Environmental Engineering*, *5*, 100173. https://doi.org/10.1016/j.cscee.2021.100173

Hao, X. D., Li, J., van Loosdrecht, M. C. M., & Li, T. Y. (2018). A sustainability-based evaluation of membrane bioreactors over conventional activated sludge processes. *Journal of Environmental Chemical Engineering*, *6*(2), 2597–2605. https://doi.org/10.1016/j.jece.2018.03.050

Henze, M., Harremoes, P., Arvin, E., & la Cour Jansen, J. (2019). *Wastewater treatment. Biological and chemical processes* (4th ed.). Springer-Verlag.

Jahn, L., Saracevic, E., Svardal, K., & Krampe, J. (2019). Anaerobic biodegradation and dewaterability of aerobic granular sludge. *Journal of Chemical Technology & Biotechnology*, *94*(9), 2908–2916. https://doi.org/10.1002/jctb.6094

Jönsson, K., Johansson, P., Christensson, M., Lee, N., Lie, E., & Welander, T. (1996). Operational factors affecting enhanced biological phosphorus removal at the waste water treatment plant in Helsingborg, Sweden. *Water Science and Technology*, *34*, 67–74. https://doi.org/10.1016/0273-1223(96)00496-9

Krampe, J., & Krauth, K. (2003). Oxygen transfer into activated sludge with high MLSS concentrations. *Water Science and Technology*, *47*(11), 297–303. https://doi.org/10.2166/wst.2003.0618

Layer, M., Villodres, M. G., Hernandez, A., Reynaert, E., Morgenroth, E., & Derlon, N. (2020). Limited simultaneous nitrification-denitrification (SND) in aerobic granular sludge systems treating municipal wastewater: Mechanisms and practical implications. *Water Research X*, *7*, 100048. https://doi.org/10.1016/j.wroa.2020.100048

Llácer-Iglesias, R. M., López-Jiménez, P. A., & Pérez-Sánchez, M. (2021). Energy self-sufficiency aiming for sustainable wastewater systems: Are all options being explored? *Sustainability*, *13*(10), 5537. https://doi.org/10.3390/su13105537

Nations, U. (2022). Sustainable Development Goals. https://sdgs.un.org/goals#goals

Pronk, M., de Kreuk, M. K., de Bruin, B., Kamminga, P., Kleerebezem, R., & van Loosdrecht, M. C. M. (2015). Full scale performance of the aerobic granular sludge process for sewage treatment. *Water Research*, *84*, 207–217. https://doi.org/10.1016/j.watres.2015.07.011

Rosso, D., Larson, L. E., & Stenstrom, M. K. (2008). Aeration of large-scale municipal wastewater treatment plants: State of the art. *Water Science and Technology*, *57*(7), 973–978. https://doi.org/10.2166/wst.2008.218

Seifert, C., Krannich, T., & Guenther, E. (2019). Gearing up sustainability thinking and reducing the bystander effect – A case study of wastewater treatment plants. *Journal of Environmental Management*, *231*, 155–165. https://doi.org/10.1016/j.jenvman.2018.09.087

Silva, C., & Rosa, M. J. (2022). A comprehensive derivation and application of reference values for benchmarking the energy performance of activated sludge wastewater treatment. *Water*, *14*(10), 1620. https://doi.org/10.3390/w14101620

Strubbe, L., Dijk, E. J. H., Deenekamp, P. J. M., Loosdrecht, M. C. M., & Volcke, E. I. P. (2023). Oxygen transfer efficiency in an aerobic granular sludge reactor: Dynamics and influencing factors of alpha. *Chemical Engineering Journal*, *452*, 139548. https://doi.org/10.1016/j.cej.2022.139548

Uthirakrishnan, U., Godvin Sharmila, V., Merrylin, J., Adish Kumar, S., Dharmadhas, J. S., Varjani, S., & Rajesh Banu, J. (2022). Current advances and future outlook on pretreatment techniques to enhance biosolids disintegration and anaerobic digestion: A critical review. *Chemosphere*, *288*, 132553. https://doi.org/10.1016/j.chemosphere.2021.132553

van Dijk, E. J., van Loosdrecht, M. C., & Pronk, M. (2021). Nitrous oxide emission from full-scale municipal aerobic granular sludge. *Water Research*, *198*, 117159. https://doi.org/10.1016/j.watres.2021.117159

van Leuven, G., Henneberger, S., & Latham, C. (2010). Theoretical and Experimental Study on Energy Efficiency of Twin Screw Blowers Compared to Rotary Lobe Blowers. http://ww1.prweb.com/prfiles/2010/06/29/4204174/0_TechnicalWhitepaperLobevsscrew.pdf

Winkler, M.-K. H., & van Loosdrecht, M. C. M. (2022). Intensifying existing urban wastewater. *Science*, *375*(6579), 377–378. https://doi.org/10.1126/science.abm3900

Yang, L., Zeng, S., Chen, J., He, M., & Yang, W. (2010). Operational energy performance assessment system of municipal wastewater treatment plants. *Water Science and Technology*, *62*(6), 1361–1370. https://doi.org/10.2166/wst.2010.394

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Ekholm, J., de Blois, M., Persson, F., Gustavsson, D. J. I., Bengtsson, S., van Erp, T., & Wilén, B.-M. (2023). Case study of aerobic granular sludge and activated sludge—Energy usage, footprint, and nutrient removal. *Water Environment Research*, *95*(8), e10914. https://doi.org/10.1002/wer.10914