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A full scale worm reactor for efficient sludge reduction by predation in a wastewater treatment plant

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ABSTRACT

Sludge predation can be an effective solution to reduce sludge production at a wastewater treatment plant. Oligochaete worms are the natural consumers of biomass in benthic layers in ecosystems. In this study the results of secondary sludge degradation by the aquatic Oligochaete worm *Aulophorus furcatus* in a 125 m^3 reactor and further sludge conversion in an anaerobic tank are presented. The system was operated over a period of 4 years at WWTP Wolvega, the Netherlands and was fed with secondary sludge from a low loaded activated sludge process. It was possible to maintain a stable and active population of the aquatic worm species *A. furcatus* during the full period. Under optimal conditions a sludge conversion of 150–200 kg TSS/d or 1.2–1.6 kg TSS/m^3/d was established in the worm reactor. The worms grew as a biofilm on carrier material in the reactor. The surface specific conversion rate reached 140–180 g TSS/m^2d and the worm biomass specific conversion rate was 0.5–1 g TSS sludge/g dry weight worms per day. The sludge reduction under optimal conditions in the worm reactor was 30–40%. The degradation by worms was an order of magnitude larger than the endogenous conversion rate of the secondary sludge. Effluent sludge from the worm reactor was stored in an anaerobic tank where methanogenic processes became apparent. It appeared that besides reducing the sludge amount, the worms' activity increased anaerobic digestibility, allowing for future optimisation of the total system by maximising sludge reduction and methane formation. In the whole system it was possible to reduce the amount of sludge by at least 65% on TSS basis. This is a much better total conversion than reported for anaerobic biodegradability of secondary sludge of 20–30% efficiency in terms of TSS reduction.

1. Introduction

The activated sludge process is worldwide the most abundantly used process to treat wastewater. The principle of organic carbon removal with this process is based on partial aerobic respiration and partial conversion of organic matter to biomass that can be separated from the treated wastewater in a settling tank. Taken into account that 40–50% of all organic carbon removed is converted to biomass (i.e. sludge) this can be regarded as an unwanted side-product of the process.

Abbreviations: COD, chemical oxygen demand; HRT, hydraulic retention time; p.e., people equivalent; SRT, solid retention time; ST, sludge tank; TSS, total suspended solids; VVM, volume of gas per volume of liquid per minute; WR, worm reactor; WWTP, waste water treatment plant.

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Nomenclature

$CO_2^{G,IN}$ [kg/m^3]	Oxygen concentration of inflow gas
CO_2^{G}[kg/m^3]	Oxygen concentration of off-gas
f_N [gN/gTSS]	Nitrogen content of organic sludge
f_d [-]	biodegradability
k [d^{-1}]	degradation rate constant
NH_4^{IN} [gN/m^3]	Ammonium concentration of inflow
NH_4 [gN/m^3]	Ammonium concentration of outflow
NO_3^{IN} [gN/m^3]	Nitrate concentration of inflow
NO_3 [gN/m^3]	Nitrate concentration of outflow
r_x [kgTSS/m^{3}d]	rate of sludge reduction
r_{sCOD} [gCOD/m^{3}d]	rate of soluble COD formation
r_{NO_3} [gN/m^{3}d]	rate of nitrate formation
r_{N_2} [gN/m^{3}d]	rate of nitrogen gas formation
t [d]	time
TSS^{IN} [kg/m^3]	solids concentration sample point 0
TSS^{OUT} [kg/m^3]	solids concentration sample point 1
ΔTSS [kg]	amount of solids degraded
TSS^{init} [kg]	initial amount of solids
V_R [m^3]	reactor volume
γ_x [gCOD/gTSS]	conversion factor for oxidation of organic carbon to CO_2
γ_{NO_3} [gCOD/gN]	conversion factor for oxidation of ammonia to nitrate
γ_{N_2} [gCOD/gN]	conversion factor for oxidation of ammonia to nitrogen gas
δ [-]	relative difference between gas measurements and dry weight based conversion rates
Φ_G [m^3/h]	gas flow
Φ_L [m^3/d]	liquid flow over worm reactor

Produced sludge in many places has to be disposed of by incineration or other costly techniques as reviewed by Fytilli and Zabaniotou (2008). Therefore sludge minimisation techniques have become a major research topic in recent years (Wei et al., 2003; Pérez-Elvira et al., 2006; Carrère et al., 2010). These techniques can be applied directly in the activated sludge process or in combination with anaerobic sludge digestion. Some proposed schemes seem to rely on decreasing sludge production by increasing the total sludge age (e.g. Cannibal, OSA). Other techniques are based on the increase of the solubilisation/hydrolysis rate and or increase the biodegradability. Used methods include mechanical, chemical, thermal and biological processes (Fig. 1) and may be placed in-line or as pre-treatment step. The drawback of most of these methods is that they are energy and cost intensive.

Most of the proposed strategies are not based on the basic cycling of organic carbon in natural systems. Complex organic carbon in nature is generally converted by higher organisms including benthic worms and it seems a logical route to incorporate worms into a system for sludge minimisation. Implementation of this ecological principle for enhanced sludge degradation in WWTP was demonstrated on laboratory scale by various researchers: Ratsak et al. (1993) described the growth of Naididae worm species in a wastewater treatment plant; Rensink et al. (1997) measured sludge reduction by Tubificidae worms in a trickling filter to be roughly 50% on TSS basis although Luxmy et al. (2001) did not measure any significant breakdown of sludge by worms. Wei et al. (2006) described a system that could be used for cultivation of Tubificidae. At about the same time Elissen et al., 2006 described a reactor concept in which *Lumbriculus variegatus* could be cultivated. Hendrickx et al. (2009) investigated this system set-up further and explored the influence of operational parameters.

While the abovementioned research established evidence that sludge conversion by aquatic worm species was possible and explored the influence of various operational parameters, a proper design for a full scale system has not been reported nor has the type of most effective benthic worm for practical conditions been investigated. In order to get insight in the relevant factors that play a role in the full scale operation of worm predation on a wastewater treatment plant a pilot system was developed. The main goal was to show the feasibility of adding a benthic worm reactor in a sludge reduction strategy and to demonstrate that stable sludge reduction rates can be established at a wastewater treatment plant.

In this study the results of conversion of secondary sludge by the aquatic Oligochaete worm *Aulophorus furcatus* in a 125 m^3 reactor and further degradation in an anaerobic tank are presented. This system was operated over a period of more than 4 years at WWTP Wolvega, the Netherlands (coordinates 52.8867N; 5.9898W, http://maps.google.nl/maps?q=52.8867,+5.9898).

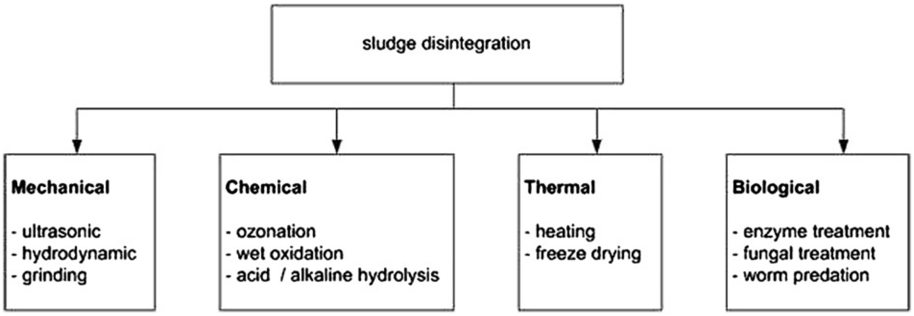

Fig. 1 – Overview of methods that aim for sludge disintegration for enhanced biodegradability.

2. Material and methods

2.1. *WWTP and sludge characteristics*

The sludge was produced in a low loaded conventional activated sludge system, using nitrification/denitrification for N removal and with phosphate precipitation with iron for P-removal. The sludge loading rate was 0.07 kg COD/kg TSS/d and the COD/N ratio of the influent was around 10:1. There was no primary settling tank present and the sludge age was approximately 20 days. The VSS/TSS ratio of the secondary sludge varied over the year between 65% and 75%.

The capacity of the WWTP was 70.000 p.e. and approximately 25% of the produced sludge was processed via the worm reactor system. The sludge was obtained from the settling tank and mixed with effluent to obtain the appropriate concentration of 1–3 g/l of TSS. Large particles were removed by a drum filter with a mesh size of 500 μm. Although large particles were selectively removed it was observed that the filter unit operation had no significant influence on the VSS/TSS ratio of the sludge. Ammonium and soluble COD concentrations in the obtained secondary were generally low with ammonium concentrations of around 1–3 mg/l and soluble COD concentrations at around 50–100 mg/l.

2.2. *Process configuration and operational conditions*

The important unit operations are the worm reactor and the anaerobic sludge sedimentation and holding tank (further referred to as sludge tank). A schematic overview of the process set-up is presented in Fig. 2. Part of the sludge was directed via the worm reactor (stream 0) to the sludge tank (stream 1). Another part was directly fed to the anaerobic tank sludge tank via a bypass stream designated 0b.

Operational parameters varied over time and 3 operational phases could be distinguished.

1. Focus on maximum degradation efficiency in the worm reactor in 2007. The worm reactor loading rate was around 2–3 kg TSS/m^3d while the sludge tank loading rate was around 0.3 kg TSS/m^3d. Bypass to the sludge tank was not active.
2. Focus on operational parameters for maximum degradation in total system (i.e. worm reactor in combination with sludge tank) in 2008. The worm reactor loading rate was around 3 kg TSS/m^3d while the sludge tank loading rate was around 0.3 kg TSS/m^3d. Around 25% of the total processed sludge was added via the bypass to the sludge tank.
3. High throughput conditions with the goal of maximizing the total degradation in terms of mass and not percentages in 2009 and 2010. The worm reactor loading rate was between 4 and 8 kg TSS/m^3d while the sludge tank loading rate was around 0.5 kg TSS/m^3d. Around 40% of the total processed sludge was added via the bypass to the sludge tank.

In the sludge tank the sludge was thickened to a concentration of 25–35 g/l of TSS and periodically transported to a central treatment facility. The water released from the sedimentation process was recirculated to the aeration tank (stream 2).

The basic of the design of the worm reactor included a carrier for immobilisation of the worms in the system, and

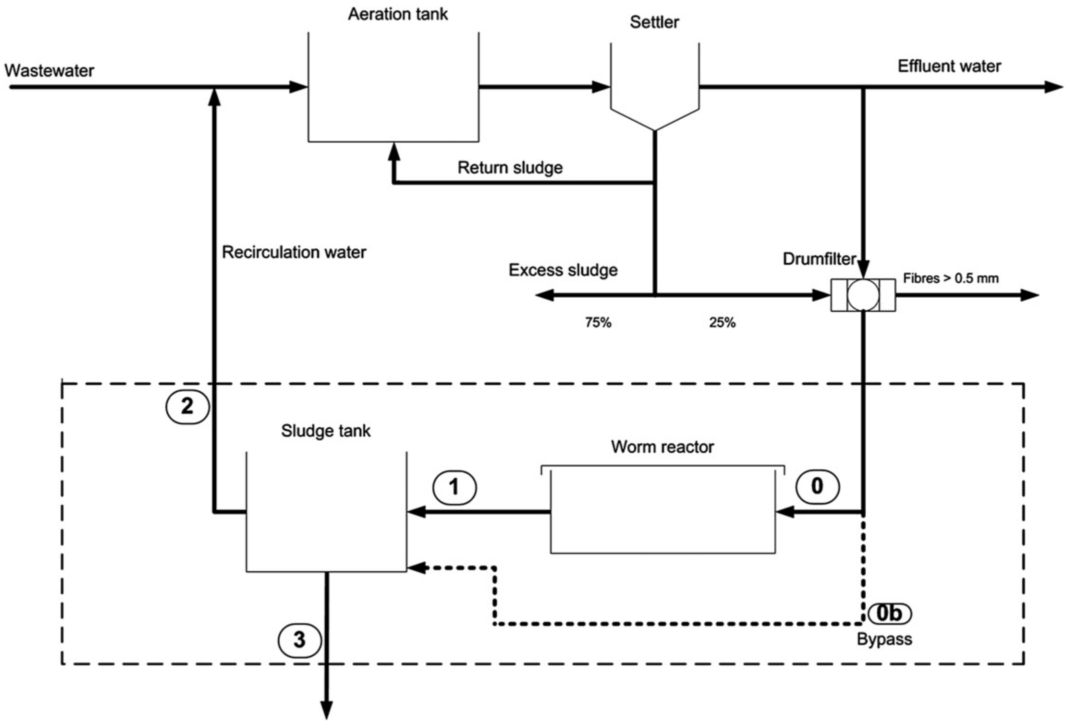

Fig. 2 – Schematic diagram of worm reactor set-up, system boundaries for this study are indicated with the dashed box. Sample points have been numbered 0–3. Unit operations of interest are the worm reactor (WR) and the sludge tank (ST).

an appropriate sludge concentration in the feed. The latter is important for maintaining an adequate dissolved oxygen concentration and preventing toxic ammonium levels. The worm reactor was designed to be able to retain *A. furcatus* worm biomass using a specific carrier material consisting of a plastic mesh of 5 mm. The reactor temperature was maintained at 25 °C using a heat exchanger and heat pump. The reactor was aerated in an airlift mode to provide appropriate mixing. Perspectives of the worm reactor are presented in Fig. 3.

It was not necessary to inoculate the reactor since *A. furcatus* appeared to be naturally present in the secondary sludge and was able to colonize the reactor with a doubling time of approximately 1 week; however at some points in time, mainly after technical maintenance of the reactor inoculations were performed to achieve quicker start-up.

The reactor was operated for 1434 days (and counting); data on sludge conversion rates was available for a period from 7th of November 2006 until 11th of October 2010. Average operational parameters (including the full period) for the worm reactor and the sludge tank are presented in Table 1.

The solid retention time (SRT) in the worm reactor was estimated to be 1–3 times higher than the hydraulic retention time (HRT) because of sludge retention by the worms on the carrier material. The SRT in the sludge tank was approximately 30 times higher than the HRT depending on the final concentration of the sludge during sedimentation. SRT was established by calculating the amount of solids present divided by the solid mass flow.

2.3. Analysis – measurements

Reactor performance and operational conditions were monitored according to the scheme presented in Table 2.

2.4. Analysis – kinetic analysis

Degradation rates were calculated from online TSS concentration and volumetric flow measurements according to Eq. (1).

$$r_X = \frac{\Phi_L}{V_R}\left(\mathrm{TSS}^{\mathrm{IN}} - \mathrm{TSS}^{\mathrm{OUT}}\right) \tag{1}$$

To investigate the contribution of sludge degradation by worms to the total degradation, the degradation rate of secondary sludge from WWTP Wolvega was determined in aerated batch experiments with no worms present. Like in the worm reactor large particles were removed from the sludge by a drum filter with a mesh size of 500 μm. Temperature was maintained at 25 ± 1 °C to ensure operational conditions comparable with the worm reactor. The obtained data was fitted to a first order model according to Eq. (2).

$$\frac{\Delta\mathrm{TSS}}{\mathrm{TSS}^{\mathrm{init}}} = f_d\left(1 - e^{k\cdot t}\right) \tag{2}$$

Fitted parameters included the biodegradable fraction (f_d) and degradation rate constant (k). These parameters were subsequently used to estimate what the extent of degradation would be in a continuous system identical to the worm reactor using Eq. (3).

$$\frac{\Delta\mathrm{TSS}}{\mathrm{TSS}^{\mathrm{init}}} = \frac{f_d\cdot\mathrm{SRT}}{\mathrm{SRT} + 1/k} \tag{3}$$

2.5. Analysis – mass balancing

Results were evaluated by on site off-gas analyses (Hellinga et al., 1996). Assuming steady-state conditions, oxygen transfer rates could be related to sludge degradation using a COD balance (Eq. (4)) while assuming nitrite concentration were negligible.

$$\frac{\Phi_G}{V_R}\left(C_{O_2}^{G,\mathrm{IN}} - C_{O_2}^{G}\right) = \gamma_X r_X - r_{\mathrm{sCOD}} + \gamma_{\mathrm{NO_3}} r_{\mathrm{NO_3}} + \gamma_{\mathrm{N_2}} r_{\mathrm{N_2}} \tag{4}$$

Rates on sludge degradation, soluble COD and nitrate formation were readily available from direct measurements. In order to quantify the rate of nitrogen gas formation that was required for evaluation of the COD balance, it was assumed that the nitrogen content of the organic fraction of the degraded sludge was 10% on mass basis (Gujer et al., 1999) and that this was the same inflow and outflow. Subsequently the nitrogen gas production rate was calculated as follows:

$$r_{\mathrm{N_2}} = f_N r_X + \frac{\Phi_L}{V_R}\left(\mathrm{NO_3^{IN}} - \mathrm{NO_3}\right) + \frac{\Phi_L}{V_R}\left(\mathrm{NH_4^{IN}} - \mathrm{NH_4}\right) \tag{5}$$

To evaluate the COD balance the relative difference between gas measurements and dry weight based conversion rates was calculated according to Eq. (6).

$$\delta = 1 - \frac{\frac{\Phi_G}{V_R}\left(C_{O_2}^{G,\mathrm{IN}} - C_{O_2}^{G}\right)}{\gamma_X r_X - r_{\mathrm{sCOD}} + \gamma_{\mathrm{NO_3}} r_{\mathrm{NO_3}} + \gamma_{\mathrm{N}} r_{\mathrm{N_2}}} \tag{6}$$

Fig. 3 – From left to right: operational worm reactor; carrier material with worm biomass; zoom in on carrier material with mesh size of 5 mm; *Aulophorus furcatus* under microscope.

Table 1 – Average worm reactor and sludge tank operational parameters.

	Worm reactor	Sludge tank	
Volume	125	900	m^3
Carrier material	1100	n/a	m^2
Volumetric flow	250	250	m^3/d
Hydraulic retention time	12	86	h
Solid retention time	[12–36]	[1500–2500]	h
Solids concentration	[1–3]	[25–35]	g/l
Temperature	[20–29]	[4–18]	°C
Dissolved oxygen	>4	<0,1	mg/l

3. Results and discussion

Over a period of 1434 days, the reactor was operated with a fluctuating but relatively stable worm activity. A relatively stable worm population was maintained with densities of on average 200 ± 90 g dry weight worms per m^2 of carrier material; the dominant species was identified as *A. furcatus* by microscopic inspection (a movie of worm biomass from the reactor is included as supplementary data available at http://homepage.tudelft.nl/8n9f7/aulophorus.avi and will be embedded in final pdf version). The worms formed a kind of biofilm that almost fully covered the carrier material. Apparently, reactor conditions favoured growth of this species since over the years *A. furcatus* remained the dominant species as was observed by microscopic inspection.

Supplementary video related to this article can be found at doi:10.1016/j.watres.2011.08.046.

3.1. *Conversion in the worm reactor*

Sludge conversion rates based on dry weight and volumetric flow measurements were determined on a daily basis and this reflected overall reactor performance in terms of sludge conversion (Fig. 4).

The sludge loading to the reactor was varying during the operational period due to changes in experimental goals. It should be noted that this included periods with lower conversion due to deliberate reactor operations that were not optimal. Under optimal conditions a sludge conversion efficiency of 150–200 kg TSS/d or 1.2–1.6 kg TSS/m^3/d or 30–40% on TSS basis could be achieved. Assuming an average VSS content of 70% the sludge reduction in the worm reactor was therewith estimated to be around 50% on VSS basis.

The worm surface specific conversion rate under these conditions was 140–180 g TSS/m^2d and the worm biomass specific conversion rates was 0.5–1 g TSS sludge/g dry weight worms per day. This biomass specific consumption rate was in the same order of magnitude as was measured for *L. variegatus* in a continuous reactor by Hendrickx et al. (2009).

In general only a weak correlation between the sludge loading and conversion rates was observed. Apparently many other factors were influencing the conversion rates, with adequate supply of sludge being just one of the boundary conditions for adequate functioning of the system. Furthermore, reactor performance was sometimes influenced by technical failures of the full scale installation: the low conversion in the last months of 2009 was due to technical problems with pump machinery.

Variations in sludge conversions could however be explained mainly by taking into account that it was not well feasible to maintain a constant high growth rate of the *Aulophorus* biomass. The worm biomass retention time was controlled by regularly manually removing a fraction of the biomass from the carrier material (this biomass leaves the reactor then via the normal effluent flow and is included in the solids mass balance). In periods when the worm biomass was controlled at a low biomass amount of around 50–100 g dry weight/m^2 (i.e. high growth rate) conversions were higher than when large amounts (300–400 g dry weight/m^2) of biomass accumulated. Considering the generally and comparatively high surface specific conversion rates of 140–180 g TSS/m^2d it is to be expected that at high biomass

Table 2 – Overview of the sampling and analysis methods.

Measurement	Sample point[a]	Frequency ($week^{-1}$)	Method	Accuracy
Volumetric flow	0,1	Online	Magnetic field	5%
TSS concentration	0,1	Online	Spectrophotometric	10%
Temperature	WR,ST	Online	Thermometer	1%
Dissolved oxygen	WR	Online	LDO	5%
Volumetric flow	3	[0,05–1]	Manual[b]	5%
TS concentration	3	[0,05–1]	Dry weight	10%
Ammonium concentration	WR,0,1,2	1	Spectrophotometric	10%
Nitrite concentration	WR,0,1,2	1	Spectrophotometric	10%
Nitrate concentration	WR,0,1,2	1	Spectrophotometric	10%
Phosphate concentration	WR,0,1,2	1	Spectrophotometric	10%
Soluble COD concentration	0,1,2	1	Spectrophotometric	10%
Worm biomass on carrier	WR	2	Manual[c]	20%
O_2 gas consumption	WR	online	Paramagnetic	5%
CO_2 gas consumption	WR	online	IR spectrometry	5%

a See process overview (Fig. 2). WR = worm reactor, ST = sludge tank.
b Transport: 36 m^3 per truck.
c Removal of biomass with a water vacuum cleaner and subsequent dry weight determination.

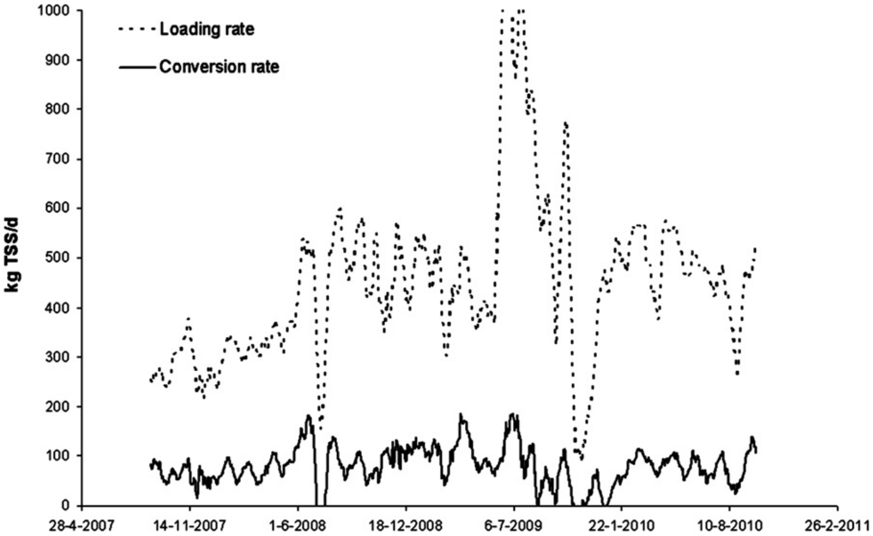

Fig. 4 – Loading and conversion rates in the worm reactor. Values are average dry weight based measurements over 15 days.

densities the inside of the biofilm becomes anaerobic causing the worm biomass to function sub-optimal.

Although ammonium and soluble COD were released during the conversion of the sludge by the worms concentration remained relatively low with ammonium concentrations around 5–20 mg/l and soluble COD concentrations at around 100–200 mg/l.

3.2. *Conversion in combination with the sludge tank*

The sludge tank was originally intended for sedimentation and storage. However, significant anaerobic degradation was observed in this stage of the process despite a low temperature (Table 3).

Results varied over the 3 distinct operational phases: in 2007 the loading rate of the system was relatively low resulting in a good conversion efficiency in the worm reactor. In this phase it was found that after predation in the worm reactor, an additional 50% of the sludge disappeared upon storage in the sludge tank. Off-gas analysis demonstrated that TSS removal was associated with conversion to methane at ambient temperature (i.e. 4–20 °C).

In the second phase in 2008 the conversion efficiency in the sludge tank was even higher while conversion in the worm reactor was lower. Results from the high throughput experiment in 2009 and 2010 indicated that conversion efficiency decreased at high loading rates while volumetric conversion rates increased.

In conclusion it was observed that it was possible to obtain 65% sludge reduction on TSS basis by application of the worm reactor in combination with the sludge tank. This is a much better conversion than reported for anaerobic digestion which typically has a degradation efficiency of no more than 20–30% on TSS basis (Li and Noike, 1992; Shimizu et al., 1993; Lin et al., 1997; Lafiitte-Trouqué et al., 2002; Valo et al., 2004; Bolzonella et al., 2005; Bougrier et al., 2006a, 2006b).

3.3. *Control experiment results*

To compare the decay of biomass by worm grazing with endogenous decay in the sludge we performed batch control experiments with the secondary sludge (Fig. 5). The aerobic endogenous decay could be described with a first order decay kinetics expression with a decay coefficient of 0.12 [d^{-1}] and a maximal degradability of 14%.

This decay coefficient is in accordance with the aerobic endogenous respiration rates typically used in the Activated Sludge Model No. 3 (Gujer et al., 1999; Koch et al., 2001) which

Table 3 – Overview of the conversion in the worm reactor and the sludge tank.

	Worm reactor			Sedimentation tank			Combined conversion
	loading rate	conversion		loading rate	conversion		
	kg/m^3/d	kg/m^3/d	%	kg/m^3/d	kg/m^3/d	%	%
2007	2.66	0.86	32%	0.34	0.16	49%	66%
2008	3.03	0.61	20%	0.39	0.21	55%	64%
2009–2010	4.37	0.65	15%	0.53	0.26	40%	49%

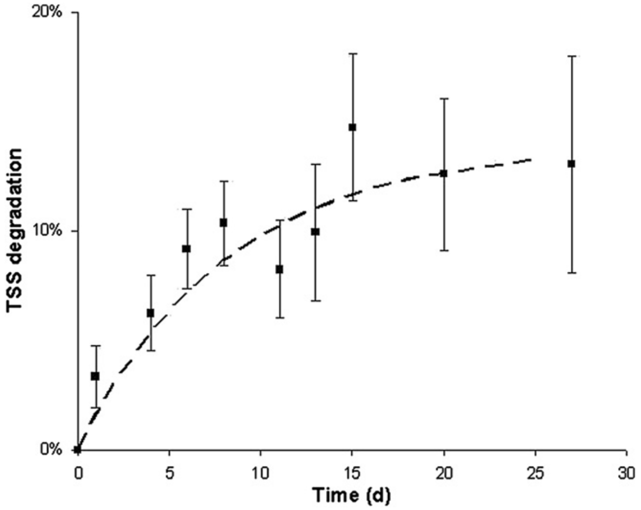

Fig. 5 – Results of batch aerobic degradation of the activated sludge from WWTP Wolvega with no worms present. Errors bars represent standard deviation of triplicate tests. Dashed line represent fitted first order model.

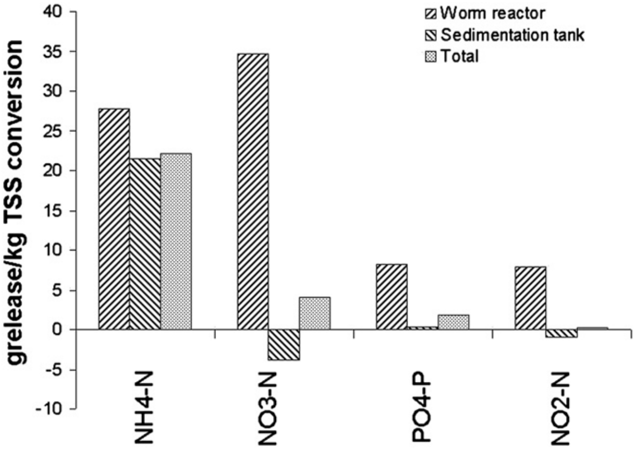

Fig. 6 – Soluble nitrogen and phosphate production per amount of TSS degraded in the worm reactor and sludge tank. Note that negative net values for nitrite and nitrate production are due to denitrification in the sludge tank.

were 0.3–0.4 [d^{-1}] at 25 °C for the active biomass. The observed value of 0.12 d^{-1} refers to the total sludge which is a mixture of active biomass, residual particalute COD and inorganic solids. Preliminary experiments on anaerobic decay of the sludge indicated a decay coefficient of 0.04 d^{-1}.

Comparison with the worm reactor using a SRT of 2 days indicates that the degradation efficiency in a continuous system without worms would be only 2% on TSS basis. Apparently the observed sludge degradation in the worm reactor was primarily caused by predation of sludge by worms leading to order of magnitude increase in the degradation rate over standard endogenous processes.

3.4. *Evaluation of nitrogen and phosphorus release*

Sludge degradation leads to nitrogen and phosphate release. In the aerated worm reactor this leads to nitrification. Denitrification occurs in the worm biofilm on the carrier material and in the sludge tank. The nitrogen and phosphorus release during sludge degradation in the experimental period is reported in Fig. 6. Measurements were only conducted on the worm reactor during an 872 days period (18-01-2008–6-8-2010) and sludge tank data were taken from a 414 days period (20-4-2009–6-8-2010).

Total nitrogen ($NH_4 + NO_2 + NO_3$) release from the worm reactor was 70 g N per kg TSS conversion. Assuming 10% (w/w) nitrogen content of the sludge this leads to the conclusion that an additional 30 g N per kg TSS conversion was potentially processed by denitrifying bacteria to produce nitrogen gas. Phosphate release was 8 g PO_4–P per kg TSS conversion in the worm reactor. Assuming a 1–2% (w/w) phosphate content of the sludge leads to indications of occurrence of precipitation reactions (e.g. $CaPO_4$, NH_4MgPO_4 or $FePO_4$); this has a very limited influence of the solids balance, but could be further investigated in future optimisation of the process.

In the sludge tank the total nitrogen release was 16 g N per kg TSS conversion, which was significantly lower than in the worm reactor. Note that not all nitrate produced in the worm reactor was subsequently converted by denitrification processes in the sludge tank: at 2 days HRT on average inflow contained 10–20 mg/l nitrate-N while outflow contained 3–10 mg/l nitrate-N, since mixing was only done sporadically for short periods this can partly be resulting from short circuiting flows in the tank. Phosphate release in the sludge tank was negligible possibly due to the high concentration of iron that was set free from the sludge under anaerobic conditions. Relatively large quantities of iron were present in the sludge since WWTP Wolvega applies chemical phosphate removal. Over the total process the net release of nutrients was approximately 30 g nitrogen and 2 g phosphorus per kg TSS conversion.

The cause of the low net formation of nitrogen and the underlying processes in the sludge tank are interesting topics for further investigation, but are for practical purposes outside the scope of this report.

3.5. *Mass balance*

Since measurement of sludge amounts at wastewater treatment plants is always cumbersome we evaluated the sludge removal also by measuring the oxygen conversion in the worm reactor. Worm reactor inflow and off-gas oxygen and CO_2 concentrations were measured over a period of 23 days and oxygen and carbon dioxide conversion was compared to the dry weight balance over the worm reactor (Fig. 7).

Average inflow oxygen gas concentration was 20.91% while average outflow gas concentration was 20.16%. The oxygen uptake rate was 1.70 kg/m^3d at a gas flow rate of 0.2 VVM (Dissolved oxygen was around 5 mg/l).

For short periods dry weight measurements indicated strong deviations from the gas measurements. It appears that the gas measurements are giving a more reliable picture of the conversions in the worm reactor while the dry weight

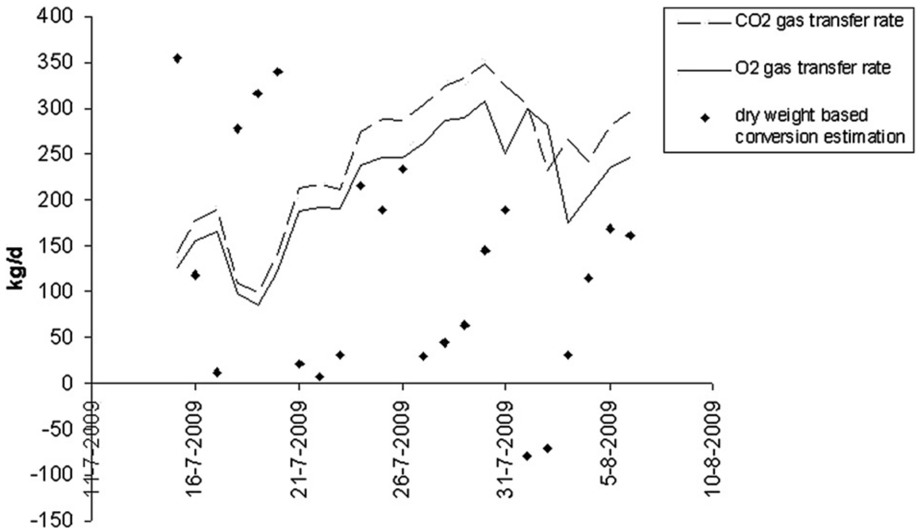

Fig. 7 – Worm reactor oxygen and CO_2 transfer rates and dry weight based conversion estimation (Eq. (1), material and methods).

balance only gives an accurate measure for the amount of sludge reduction over longer periods. Variance in the dry weight based conversion estimation was primarily caused by accumulation effects of sludge in the worm biofilm in the reactor.

This trend was evaluated by comparing oxygen gas transfer to dry weight based conversion estimation, nitrification and denitrification rates (using the Eq. (2), material and methods). The used nitrate production rate was 35 mg/g sludge conversion and the denitrification rate (obtained from the overall N-balance was) 30 mg/g sludge conversion while nitrite production was negligible; based on evaluation of the nitrogen release (Fig. 6). The relative error in this was subsequently calculated (using Eq. (4), material and methods) for periods of different lengths and plotted in Fig. 8.

These results indicate that measurements of reduction based on dry weight were only representative over longer periods. For establishing the amount of sludge degradation, periods of at least 2–3 week need to be evaluated if based on dry weight measurements.

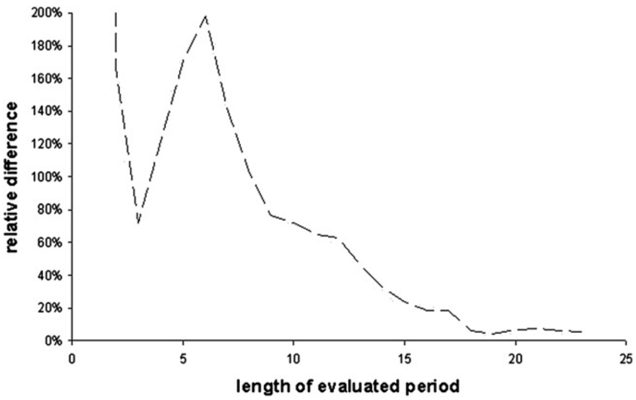

Fig. 8 – Relative difference between the cumulative sludge conversion estimated from dry weight measurements and gas transfer measurements. *X*-axis represents the length of the evaluated period.

3.6. *General evaluation*

This work shows that use of benthic worms can greatly reduce sludge production. Compared to conventional reported values for secondary sludge digestion the sludge reduction was roughly doubled. Learning from nature and adapting this in engineered systems is a well known approach in wastewater treatment that can be extended to sludge treatment. It was possible to maintain a relatively stable worm population in a simple reactor design. Regarding economic potential, the proposed system with a self-organizing worm biofilm in a well mixed reactor will be favoured over more complex designs proposed in literature (i.e. Elissen et al., 2006; Hendrickx et al., 2009).

There is still room for improvement of this system. We observed that worms have a high biomass specific consumption rate at low worm biomass densities. Having a good worm biomass control in the process will therefore improve the efficiency of the process. Conversion in the worm reactor appears to be limited by the maximum oxygen transfer rate from the bulk liquid to the worm biofilm on the carrier material.

An interesting finding was that while in the worm reactor the sludge removal was in the same order as for digestion systems on secondary sludge, there was still a sludge degradation in the subsequent anaerobic sludge holding tank. This anaerobic sludge digestion occurred under psychrophilic conditions. This was not anticipated beforehand and therefore not optimised. It appeared as if the worms formed a kind of pre-treatment system not only oxidising sludge but also converting it in a better biodegradable form similar as reported for e.g. ozone treatment or physical processes. Possibly, the mechanism by which worms degrade sludge inside their body works also outside of their body. Further investigation is needed to clarify the working of this mechanism.

Be that as it may, the observed phenomenon opens the option to optimise the total system by minimising the conversion in the worm reactor and maximising the anaerobic digestion. Another option might be reversal of the order of the unit operations (i.e. the worm reactor and the anaerobic digestion) in order to obtain a maximal gas production. This way all easily degradable material can be converted into biogas, while relatively slow degradable material is fed to worms.

4. Conclusion

1. It was possible to maintain a stable and active population of the aquatic worm species *A. furcatus* in a full scale reactor for a period of 4 years.
2. A reduction of 65% on TSS basis was realised during this period resulting from worm predation and subsequent digestion, whereby the predated sludge digested spontaneously under psychrophilic anaerobic conditions. Overall biodegradability increased and was much higher than total conversion as reported in literature for anaerobic digestion of secondary sludge
3. Reported observations reflect the results of a large scale demonstration plant that was not optimized. It is believed that this is the first step towards a completely new and successful approach to decrease waste activated sludge combined with anaerobic digestion of the sludge.

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