IWA PUBLISHING
the International
water association

Record not found.

Sewage Treatment Plants: Economic Evaluation of Innovative Technologies for Energy Efficiency
Editor(s): Konstantinos P. Tsagarakis and Katerina Stamatelatou
Publication Date: 15 Apr 2015 • ISBN: 9781780405018
Pages: 376 • Hardback

○ Price: £ 115.00 / US$ 207.00 / € 155.25

○ IWA members price: £ 86.25 / US$ 155.25 / € 116.44

Sewage Treatment Plants: Economic Evaluation of Innovative Technologies for Energy Efficiency aims to show how cost saving can be achieved in sewage treatment plants through implementation of novel, energy efficient technologies or modification of the conventional, energy demanding treatment facilities towards the concept of energy streamlining.

The book brings together knowledge from Engineering, Economics, Utility Management and Practice and helps to provide a better understanding of the real economic value with methodologies and practices about innovative energy technologies and policies in sewage treatment plants.

Table of contents
The principles of economic evaluation and cost benefit analysis implemented in sewage treatment plants; Economic evaluation of innovative technologies aiming to increase the energy efficiency of the sewage treatment plants; Instrumentation, monitoring and real-time control strategies for efficient sewage treatment plant operation; Process integration to improve carbon, nitrogen and phosphorus removal with less aeration requirements and less sludge production; Bioreactor development with less aeration requirement; Improvement of anaerobic digestion of sewage wastewater and sludge; Development of microbial fuel cells for electricity production from sewage; Nutrient recovery from sewage; Cost saving management strategies or policies; Economic evaluation of innovative technologies aiming to increase the energy efficiency of the sewage treatment plants: Case studies.

This title belongs to **Integrated Environmental Technology Series**
Print page | **Close window**

Chapter 16

Short-cut enhanced nutrient removal from anaerobic supernatants: Pilot scale results and full scale development of the S.C.E.N.A. process

Daniele Renzi[1], Stefano Longo[2], Nicola Frison[2], Simos Malamis[2], Evina Katsou[2] and Francesco Fatone[2,]*

[1]*Alto Trevigiano Servizi srl, Via Schiavonesca Priula, Montebelluna – Treviso (Italy)*

[2]*Department of Biotechnology, University of Verona, Via Le Grazie 15, Verona – Italy*

*francesco.fatone@univr.it – Tel/Fax: +39 045 8027965

16.1 INTRODUCTION

Enhanced nutrient removal in municipal wastewater treatment plants (WWTPs) can be partly and efficiently carried out by treating the ammonium and phosphorus-rich reject water (other terms for reject water are 'return liquor', 'digester supernatant' or 'sludge digester liquid') produced from the dewatering of anaerobic digested sludge in order to meet more stringent effluent standards. In conventional plants the nitrogen flow from the reject water constitutes 10–30% of the total N-load (Cervantes, 2009; Gustavsson *et al.* 2011). As far as phosphorus is concerned, the concentration in reject water can be up to 130 mg L^{-1} (Oleszkiewicz & Barnard, 2006; Pitman, 1999; Ivanov *et al.* 2008). High P concentrations may be reached when anaerobic co-digestion of sewage sludge and organic waste are applied (Malamis *et al.* 2014; Battistoni *et al.* 2005). Thus, the reject water is returned to the activated sludge process and accounts for 10 to 50% of the nutrients in the main stream of municipal WWTPs.

In addition, innovative schemes aiming at energy neutral-positive municipal WWTPs consider the anaerobic digestion as the core process for biogas recovery from sewage sludge. As a consequence, the enhanced nutrient removal from the

digester supernatant is proposed to take place separately from the main stream and becomes a significant stage within the new-conceived WWTPs.

16.1.1 Removal or recovery?

Separate treatment of the digester supernatant requires a minimum extension because of the high temperature of the supernatant (25–35°C), potentially leading to short sludge retention times (SRTs) and high reaction rates. Furthermore, a high ammonium concentration and a low COD:N ratio favor high autotrophic ammonium reduction rates. The alkalinity content is often around 1.1 mol HCO_3 per mol NH_4^+-N, while 1.98 mol HCO_3 per mol NH_4^+-N is required for complete nitrification, so extra alkalinity is required if complete ammonium oxidation is needed. Among the different treatment technologies for digester supernatant (cfr Chapter 5), the innovative biological processes have been proved to be the most economically sustainable in terms of nitrogen removal. Compared to the physicochemical processes, these processes do not allow nitrogen recovery. However, the sustainability of the currently available and future nitrogen removal systems has been investigated by several authors (Mulder, 2003; STOWA, 2012). Ammonium can be stripped from ammonium rich side streams (e.g., rejection water) by means of air stripping. This is a well-known technique. In order to strip ammonium a high pH is required (pH 10 to 12). Usually NaOH or Ca(OH) are added as alkali to realize this pH increase. In the air stripping process the rejection water is led through a stripping column in reverse flow through an air stream. The ammonia is transferred to the air stream which is led to an absorber. The adsorbed substance contains acid (H_2SO_4 or HNO_3) in which the ammonia dissolves and ammonium salts are formed. The ammonium salts are drained from the absorber while the ammonia free air can be recycled to the stripper (STOWA, 2012).

In general, the energy demand of such a reference stripping varies from 100 to 150 MJ/kg N (aeration, heat, chemicals as well as their cost) and is significantly higher than the energy demand of the nitrogen producing Haber-Bosch process combined with Anammox (total 60 MJ/kg N).

This shows that nitrogen recovery is more expensive (1.9–3.2 €/kg N) than nitrogen removal using Anammox (0.8 €/kg N). Because of the higher energy utilization as well as, the price and quantity of the chemicals required (NaOH or CaO and H_2SO).

In contrast to nitrogen, phosphorus is a limited resource which must be recovered and reused. It is estimated that the remaining accessible reserves of phosphate rock will run out in 50 years, if the growth of demand for fertilizers remains at 3% per year (Gilbert, 2009; Elser & Bennet, 2011). Reducing usage will help the reserves last longer, but the biggest gains will probably be derived from the recovery of phosphates, both from wastewaters and livestock waste (Gilbert, 2009).

Struvite ($MgNH_4PO_4 \cdot 6H_2O$) is generally considered as the optimal phosphate mineral for recovery as it contains 51.8% of P_2O_5 (based on $MgNH_4\ PO_2$) and

could potentially be used as a slow-release fertilizer. If the economic and life cycle costs are taken into account, however, phosphate recovery in the form of struvite may not be the best approach, for the following reasons: (1) production of P-mineral with a high content of struvite from real wastewater is a difficult and costly process; and (2) struvite is not superior to other phosphate based compounds in terms of fertilization efficiency. Hence, phosphate recovery could be aimed at any forms of phosphate-based compounds acceptable by the fertilizer industry, depending on the onsite economic and environmental circumstances, including the local regulations about recovered material. Accordingly, efforts should also be targeted towards the use of (composted) sludge for effective fertilization (Hao *et al.* 2013). The application of innovative short-cut nitrogen removal and via-nitrite enhanced phosphorus removal can optimize the treatment costs and resource recovery, taking into account the potential usability of the recovered materials.

16.2 SHORT-CUT NITROGEN REMOVAL AND VIA-NITRITE ENHANCED PHOSPHORUS BIOACCUMULATION: FUNDAMENTALS

Short-cut nitrogen removal (SCNR) through the ammonium oxidation to nitrite and its subsequent reduction to gaseous nitrogen has gained increasing attention over the last years. The adoption of nitritation/denitritation as opposed to conventional nitrification/denitrification has significant advantages, since it theoretically reduces the oxygen demand up to 25% and requires up to 40% less organic carbon source. Furthermore, it decreases sludge production by 20–40% and carbon dioxide emissions by 20% (Gustavsson, 2010). The completely autotrophic nitrogen removal process is even more economically attractive as it further reduces energy demand and has no external carbon source requirements. However, its operational and environmental sensitivity and the facts that neither enhanced biological phosphorus removal can be achieved nor nitrogen is completely removed are major drawbacks for its potential implementation (Malamis *et al.* 2014).

To accomplish SCNR the growth of ammonium oxidizing bacteria (AOB) must be favored against the growth of nitrite oxidizing bacteria (NOB). NOB can be inhibited/washed out by maintaining a significant free ammonia (FA) and/or free nitrous acid (FNA) concentration in the reactor (FA > 1 $mgNH_3 \cdot L^{-1}$, FNA > 0.02 $mgHNO_2\text{-}N \cdot L^{-1}$) (Anthonisen *et al.* 1976; Gu *et al.* 2012). AOB are also favored against NOB at alkaline pH (8–8.8) (Zhang *et al.* 2007), high temperature (>25°C) (Hellinga *et al.* 1998) and low dissolved oxygen (DO) concentration (0.3–1.5 $mg \cdot L^{-1}$) (Peng & Zhu, 2006; Blackburne *et al.* 2008).

SCNR has been examined for strongly nitrogenous effluents, such as landfill leachate, supernatant produced from the anaerobic digestion of activated sludge (i.e., reject water) and from the organic fraction of municipal solid waste (OFMSW) (Hellinga *et al.* 1998; Fux *et al.* 2006; Ganigué *et al.* 2007; Ganigué

et al. 2012) as well as from low strength effluents such as domestic wastewater (Blackburne *et al.* 2008). Nitritation/denitritation has been examined for various environmental and operating conditions, including low and high DO concentrations in the reactor (Pollice *et al.* 2002; Blackburne *et al.* 2008; Guo *et al.* 2009a), high temperature (Hellinga *et al.* 1998), high salinity (Ye *et al.* 2009), different FA and free nitrous acid (FNA) concentrations (Park *et al.* 2010), different solids retention times (SRTs) (Pollice *et al.* 2002).

SCNR can be conveniently coupled with suitable bioprocesses for phosphorus removal through its accumulation in biomass and can be a sustainable option resulting in the production of high added value products. In fact, the mechanism of phosphorus uptake can be realized under anoxic conditions by denitrifying phosphorus accumulating organisms (PAOs) that can utilize nitrate or nitrite as electron acceptors (Kishida *et al.* 2006; Carvalho *et al.* 2007). Denitrifying PAOs require less carbon source compared to aerobic PAOs (Li *et al.* 2011; Peng *et al.* 2011). The rate of phosphate uptake can be higher in the presence of nitrite compared to nitrate (Lee *et al.* 2001).

16.3 CAPITAL AND OPERATING COST OF ANAEROBIC SIDESTREAM TREATMENT

16.3.1 Energy consumptions and costs of short-cut nitrogen removal from anaerobic sidestream

In conventional nitrification-denitrification in the main line of WWTPs, the electrical energy consumption is normally 3.5–5.7 kWh/kg $N_{eliminated}$ (Beier *et al.* 2008). Electrical energy consumption at the DEMON in Strass WWTP was reported to be 1.16 kWh/kg N (Wett, 2007) and in the case of the SBRs in Zürich, similar figures were reported: 1.0 kWh/kg N (Joss *et al.* 2009). The designed electric energy consumption for the DeAmmon in Himmerfjärden WWTP was 2.3 kWh/kg $N_{eliminated}$ (Gustavsson, 2010), which is close to the consumption for an SBR with nitritation-denitritation at Sjölunda WWTP in Sweden, 2.9 kWh/kg N (Gustavsson *et al.* 2011). The electrical energy consumption in the DeAmmon in Hattingen was reported to be as high as 5.6 kWh/kg $N_{eliminated}$ (Jardin *et al.* 2006) and was explained by the low ammonium load (Gustavsson, 2010).

Personnel requirements were estimated to be 0.25 man/year for the DeAmmon in Hattingen (Jardin *et al.* 2006). In all the marketing nitritation-anammox solutions some license or royalties costs are required. There should also be certain agreements on support, particularly in the event of failure.

Investments costs are very site-specific. The investment costs for a Sharon/Anammox installation with a capacity of 1,200 kg NH_4-N/day are estimated at €2 million (2001). The operating costs are linked to the costs for energy, methanol, and lye. In the Table 16.1 indicative economical parameters are given for a full-scale Sharon-Anammox process (van Dongen *et al.* 2001), where the prices refer

to the year 2001 and to design guidelines and technical data given by the authors van Dongen *et al*. (2001).

Table 16.1 Full scale cost of Sharon-AnAmmOx process (van Dongen *et al.* 2001).

Parameter	Unit	Case 1	Case 2
N-load	kg N/d	1200	1200
Flow rate	m³/d	2400	1000
NH$_4$-conc.	m³/d	500	1200
Investment	€ (x1.000)	2260	1810
Depreciation	€/year (x1.000)	240	196
Maintenance	€/year (x1.000)	46	41
Personnel	€/year (x1.000)	11	11
Electricity	€/year (x1.000)	82	76
Total cost	€/year (x1.000)	374	325
Cost per kg N$_{removed}$	€	1.05	0.90

On the other hand, Siegrist *et al*. (2012) reported the cost of ammonia stripping versus Nitritation/Anammox in SBR pointing out the 50% cost savings using the biological processes (Table 16.2).

Table 16.2 Cost comparison ammonia stripping versus Nitritation/Anammox in SBR (Siergrist *et al.* 2012).

	NH3-Stripping WWTP Opfikon 19.2 t NH$_4$–N$_{elim}$/year	Nitritation/Anammox WWTP St. Gallen-Au 46.6 t NH$_4$–N$_{elim}$/year
Operating costs (chemicals, energy, sludge disposal)	2.50	0.60
Maintenance costs (spare parts)	1.50	0.20
Personnel costs (25–30% of site)	1.50	0.70
Proceeds of sale of fertilizer	0.60	–
Capital cost	3.50	2.70
Net cost	8.40	4.20

Volcke *et al*. (2007) reported economic evaluations on the basis of the operating cost index (OCI) for a simulated WWTP treating 21100 m³/day, which resulted in reject water of 172 m³/day (Table 16.3).

As opposed to nitrogen, phosphorus is a non-renewable resource. There is a wide range of technologies to remove and recover phosphorus from wastewater,

including chemical precipitation, biological phosphorus removal, crystallization, novel chemical precipitation approaches and other wastewater and sludge-based methods (Morse *et al.* 1998).

Table 16.3 Economic evaluations of reject water treatment by SHARON-AnAmmOx (Volcke *et al.* 2007).

Costs (€/year)	No reject water treatment (BSM2)	Reject water treatment with SHARON-AnAmmOx
Effluent quality (EQ)	437400	359700
Aeration energy (AE)	194330	194530
Mixing energy (ME)	16200	16570
Pumping energy (PE)	57770	66620
Sludge production (SP)	239000	231000
External carbon addition (EC)	43900	2000
Methane production (MP)	−128800	−123000
TOTAL (=OCI)	859800	747400

Struvite crystallization (SC) is the most widespread. The SC process removes nitrogen and phosphorus from nutrient-rich wastewater by binding these two compounds together in the form of crystallized struvite which can be used as a slow-release fertilizer and can have commercial value depending on the local regulations on the recovered products. The theoretical composition of the so-called MAP (Magnesium-Ammonium-Phosphate) on a weight basis is 9.9% magnesium, 5.7% nitrogen, 12.6% phosphorus with the remainder being crystalline water. MAP satisfies a need for mineral slow-release fertilizers and has many potential uses in horticulture, for nurseries, golf courses, and so on. MAP is likely to be of most benefit to customers as a 'boutique' fertilizer. An alternative to supplying the product directly to end-users is to sell it in bulk to a fertilizer manufacturer for use as a raw ingredient in their products (Munch *et al.* 2001).

The sales price for the MAP has a significant impact on the economics of operating the SC Process. Until a separate business plan for the MAP has been completed, a sales price has to be estimated. Based on the nitrogen and phosphorus content alone, a sales price of $234/t is possible. This would be the price if the MAP was to be used in broad-scale agriculture. However, the intention is to use MAP as a 'boutique' fertilizer for specialized applications. For these applications, much higher sale prices can be achieved. In Japan, a sale price of $3800/t has been reported (Taruya *et al.* 2000). Dockhorn (2009) reported a price of struvite of 760 €/t MAP (6 €/kg P).

A business plan for the Australian market reported the operating cost of the SC process and concluded the following: Expenditure is mainly centered on capital

acquisitions (of SC Process units of 200–400 k€), the actual costs of which may be less than the current estimates used in these calculations, significantly increasing profits returned: (a) Growth in net asset value from zero to $3.8 million in 5 years; (b) High returns on equity of 44% in Year 5; (c) Profits returned in Year 2, profits consistently increasing to $1.5 million in Year 5; (d) Most of the equity is held in physical assets.

However, besides the reasons concerning the agronomic properties and the cost stated above (Hao *et al.* 2013), struvite is not easily marketable due to legislation constraints in some EU countries which can influence a lot the economical sustainability of the scheme anammox + struvite.

16.4 S.C.E.N.A. SYSTEM

16.4.1 Pilot-scale results

16.4.1.1 S.C.E.N.A. system integrated in co-digestion of WAS and OFMSW for bio-hythane production

The first pilot scale S.C.E.N.A. (Short-Cut Enhanced Nutrients Abatement) system was applied and validated for the treatment of the supernatant of anaerobic co-digestion of sewage sludge and organic fraction of the municipal solid waste (OFMSW) (Fatone *et al.* 2011) within the pilot hall of the Treviso (northern Italy) municipal treatment plant. The authors discussed the best available start-up strategy and carbon source to enhance the short-cut nitrogen removal and via-nitrite enhanced biological phosphorus uptake from anaerobic supernatant (Frison *et al.* 2012, 2013). The first integrated scheme of (1) two-phase anaerobic digestion for the bio-hythane production (Cavinato *et al.* 2013) and (2) via-nitrite biological nutrients removal was proposed by Frison *et al.* (2013) (Figure 16.1, adapted from Malamis *et al.* 2013).
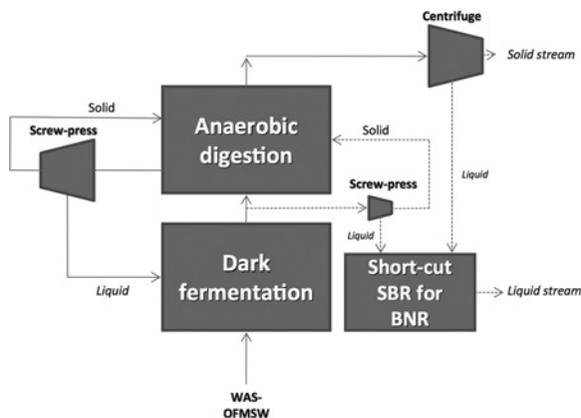
Figure 16.1 Biohythane production and scSBR integration in an anaerobic co-digestion plant (adapted from Malamis *et al.* 2013).

In this first scheme the hydrolysis reactor (dark fermentation) of the two-phase anaerobic digestion is used to provide short chain volatile fatty acids to the anoxic phase in the short-cut sequencing batch reactor (scSBR), which is treating the anaerobic supernatant. Starting from the conventional activated sludge inoculum, the start-up of the scSBR is carried out in two periods (Frison *et al.* 2012) and the stable via-nitrite route is achieved in 15–30 days. Then, nitritation-denitritation and significant phosphorus luxury anoxic uptake was observed. However, the dark acid fermentation did not optimize the contents of propionic and butyric acid that can enhance the via-nitrite (anoxic) biological phosphorus uptake.

On the basis of pilot scale trials, Frison *et al.* (2013) calculated the specific costs of via-nitrite nitrogen removal (Table 16.4). It was found that using the OFMSW fermentation liquid instead of methanol, the overall specific cost for nitrogen removal in the nitritation-denitritation decreased by 22%. In addition, the enhanced phosphorus biological removal was an important added value of the scSBR. Moreover, the added value of the contemporary via-nitrite anoxic phosphorus uptake was not considered, thus underestimating the advantages of the S.C.E.N.A. process with comparison to the complete autotrophic nitrogen removal which must be followed by struvite recovery to achieve the same nutrients removal from anaerobic supernatant.

Table 16.4 Specific costs comparison of nitritation with heterotrophic/autotrophic denitritation (adapted from Frison *et al.* 2013).

Process option	Type of carbon source	Specific costs
	–	$€ \text{ kg N}^{-1}_{removed}$
Two reactor nitritation-anammox	–	2.5
One reactor nitritation-anammox	–	2.3
DEMON® full scale		
One reactor nitritation/denitritation	Methanol	3.24–3.64
One reactor nitritation/denitritation	OFMSW liquid fermentation	2.85

16.4.2 S.C.E.N.A. system integrated in conventional treatment of sewage sludge

The S.C.E.N.A. system (Figures 16.2 and 16.3) was applied at the conventional municipal WWTP of Carbonera (Italy), where a best available carbon source (BACS) was produced from the fermentation of sewage sludge. Therefore, it was called 'Best', for the enhancement potential on nutrient removal, 'Available' because it was recovered on-site from an available waste stream.

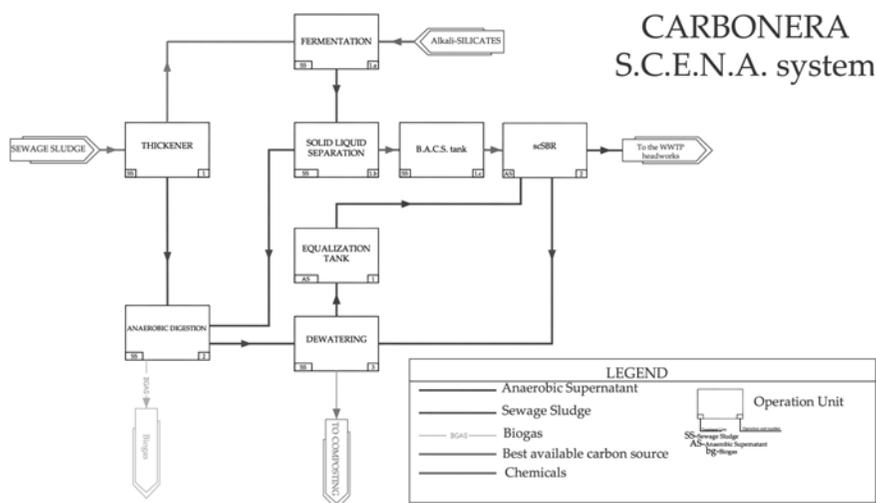**Figure 16.2** Pilot-scale S.C.E.N.A. system.

Figure 16.3 The S.C.E.N.A. system integrated in Carbonera municipal WWTP.

The S.C.E.N.A. system in municipal WWTP can be described according to the following key processes: (1) alkaline production of BACS from sewage sludge (or OFMSW); (2) nitritation in aerobic conditions (so as to also minimize N_2O emissions); (3) denitritation and via-nitrite biological phosphorus uptake achieved through the BACS dosage.

The biowaste derived alkaline fermentation liquid is rich in propionic and butyric acid and has been found to be a high added value carbon source. Several studies demonstrated that using sludge-derived Short Chain Volatile Fatty Acids (SCVFA) resulted in superior BNR performance than using synthetic acetate

(Tong *et al.* 2007; Zheng *et al.* 2010). Higher phosphorus removal efficiency was achieved with the use of SCVFA derived from WAS compared to acetate (Tong *et al.* 2007). The authors explained that the presence of propionate was probably the reason for better phosphorus removal, while the higher nitrogen removal efficiency might be due to the better use of exogenous denitrification pathway for nitrogen removal.

In the sludge fermentation process, the pH plays an important role on the hydrolysis of sludge and the production of SCVFAs from excess sludge in fermentation. Under alkaline conditions, the yield of SCVFAs can be significantly enhanced (Yuan & Weng, 2006). Recent studies have demonstrated enhanced SCVFAs production and inhibition of methanogenic activity (resulting in less SCVFAs consumption) under alkaline conditions (Wu *et al.* 2010). NaOH and $Ca(OH)_2$ are widely used for alkaline sludge treatment; the type of chemical that is used impacts on waste activated sludge (WAS) hydrolysis, acidification and dewatering ability (Kim *et al.* 2003). Thus, the type of reagent that is used for the pH adjustment in alkaline fermentation influences the effectiveness of the process. The optimum pH range 9–11 was reached using NaOH and $Ca(OH)_2$ (Su *et al.* 2013). This technique is not economically and environmentally sustainable and enhances the salinity of the carbon source, thus decreasing the rates of nitritation/nitrification. Furthermore, the sludge dewatering characteristics and the separation of the produced fermentation liquid from sludge can be adversely affected from the use of NaOH (Su *et al.* 2013; Longo *et al.* 2014). Recent studies have shown that the use of WAS fermented liquid as carbon source results in the reduction of nitrous oxide (N_2O) and nitric oxide (NO) production during the via nitrite processes (Zhu & Chen, 2011).

Therefore, the initial S.C.E.N.A. process was upgraded for application in a conventional municipal wastewater treatment plant and applied in the Carbonera WWTP.

The Carbonera (Veneto Region – Italy) WWTP plant has actual treatment potential of approximately 40,000 PE. The full scale wastewater treatment line is composed of the following operation units: screening and degritting, primary sedimentation, activated sludge process (+chemical P precipitation) and secondary clarifier; anaerobic digestion of the sewage sludge and dewatering (by centrifuge).

Currently, an S.C.E.N.A. pilot system is operating for the separate treatment of part of the reject water of the plant. The pilot unit consists of three main subunits: the alkaline fermentation unit, the membrane unit for the solid/liquid separation of the fermentation effluent and the sequencing batch reactor (SBR) for the via nitrite nutrient removal. The pilot scale fermentation unit (reaction volume 0.5 m³) receives mixed (primary and secondary sewage sludge) from the full scale WWTP plant. The sewage sludge is fermented to produce an effluent that is rich is SCVFAs. An ultrafiltration (UF) membrane filtration skid is employed for the solid/liquid separation of the fermentation effluent (MO P13U 1 m, Berghof, Germany). Fermented sludge is first screened through 50 mm to prevent clogging of the membrane modules. The sludge fermentation liquid is then directed to

the short-cut via nitrite SBR process (3 m^3) that treats separately the anaerobic supernatant, removing N and P via nitrite.

According to the pilot-scale results, the best parameters for the alkaline fermentation of sewage sludge, the solid-liquid separation and the via-nitrite nutrients removal were found out (Longo *et al.* 2015). The production of SCVFA by alkaline fermentation has proved to be highly dependent on pH and temperature. The use of wollastonite was tested in order to avoid the addition of chemicals in the alkaline fermentation process. The fermentation liquid consisted mainly of acetic, propionic and butyric acid (37, 34 and 15% respectively). Under the presence of acids, the following silicate reaction can occur

$$CaSiO_3(s) + 2H^+(aq) \rightarrow Ca^{2+}(aq) + SiO_2(s) + H_2O$$

(16.1)

Through the H$^+$ consuming reaction, it is possible to maintain an alkaline pH (8–9). Besides, an increasing pH also shifts the carbonate equilibrium towards HCO^{3-} and CO$_2^{3-}$ resulting in the precipitation of secondary carbonates as CaCO$_3$ and Ca$_3$(PO$_4$)$_2$.

The solid liquid separation is often the bottleneck of the process. The filtration performance of the fermented sludge by membrane ultrafiltration was analyzed. The addition of wollastonite decreased the CST and TTF (by 51% and 59% respectively), resulting in more favorable dewatering potentials.

The fermentation liquid produced was tested as a carbon source for nutrient removal into scSBR. The average influent, effluent characteristics together with the nutrient removal efficiency measured are reported in Table 16.5 while the nutrient removal rates are reported in Table 16.6.

Table 16.5 Average influent, effluent characteristics and nutrient removal efficiency.

Parameter	Influent (mg · L^{-1})	Effluent (mg · L^{-1})
TKN	480–520	26–50
NH$_4$-N	470–510	20–35
NO$_2$-N	<0.5	4–25
NO$_3$-N	<0.5	<0.5
PO$_4$-P	50–70	10–25
TP	80–100	15–25

Table 16.6 Specific nutrient removal rates.

sAUR (mgN/gVSS∗h)	10–15
sNUR$_{BACS}$ (mgN/gVSS∗h)	45–70
sPUR (mgP/gVSS∗h)	4.5–8

The demonstration S.C.E.N.A. system was designed according to the best parameters derived from the pilot-scale experimentation for the alkaline fermentation of sewage sludge, the solid-liquid separation and the via-nitrite nutrients removal.

Considering a design treatment potential of 40,000 PE the summary of OPEX were preliminary outlined as in Table 16.7.

Table 16.7 Estimation of annual cost for reject water treatment in Carbonera WWTP.

		C.A.S.P.*	S.C.E.N.A.
ELECTICAL ENERGY	€/year	28,000	13,300
SLUDGE DISPOSAL	€/year	26,800	19,300
PolyAlluminiumChloride	€/year	4100	–
WOLLASTONITE	€/year	–	800
PERSONNEL ANNUAL COST	€/year	700	1700

*Current conventional activated sludge process.

As far as the environmental impact is considered, the S.C.E.N.A. system allows the biological uptake of the phosphorus in a form that can be recovered after composting of the S.C.E.N.A. sludge. The impact of the S.C.E.N.A. system was also preliminary evaluated in terms of quality of the secondary effluent as shown in Table 16.8.

Table 16.8 Quality of the secondary effluent in current and simulated future scenario.

	TSS ($mg \cdot L^{-1}$)	COD ($mg \cdot L^{-1}$)	NH_4–N ($mg \cdot L^{-1}$)	NO_3–N ($mg \cdot L^{-1}$)	TP ($mg \cdot L^{-1}$)	TN ($mg \cdot L^{-1}$)
Current scenario (without S.C.E.N.A.)	5.1	4.45	1.56	7.31	2.83	8.87
Future scenario (with S.C.E.N.A.)	5.04	4.67	1.33	3.59	2.32	4.92

16.5 CONCLUSIONS

Sludge reject water is a nutrient-rich flux which should be properly treated managed for the technical, economical and environmental optimization of the nitrogen removal and phosphorus recovery in WWTPs. The completely autotrophic nitrogen removal is the most attractive biological process for the treatment of sludge reject waters in municipal WWTPs with several full scale applications. This solution cannot enhance the phosphorus bioaccumulation and should be followed by struvite crystallization for sustainable phosphorus recovery.

On the other hand, efforts for the technical, economical and environmental sustainability of wastewater treatment plants should also address the novel denitrifying biological phosphorus removal via nitrite which offers the potential to integrate phosphorus and nitrogen removal in a robust single bioreactor in which ammonium is oxidized to nitrite under aerobic conditions, while under anoxic conditions the denitrification via nitrite and enhanced biological phosphorus uptake occur simultaneously by the denitrifying phosphorus accumulating organisms. Thus, the phosphorus could be recovered via the composted sludge.

The S.C.E.N.A. system achieved these objectives in the wastewater treatment plant of Carbonera (Veneto region, northern Italy). Here the real anaerobic supernatant was treated in a nitritation-denitritation short-cut SBR, where the best available carbon source to enhance the phosphorus bioaccumulation was in-situ recovered from the sewage sludge.

The OPEX estimation proved the economic viability of the system which led the water utility Alto Trevigiano Servizi srl to apply the S.C.E.N.A. system by retrofitting an existing tank. This will also minimize the CAPEX and demonstrate how this system can be applied to integrate existing WWTPs.

16.6 REFERENCES

Anthonisen A. C., Loehr R. C., Prakasam T. B. and Srinath E. G. (1976). Inhibition of nitrification by ammonia and nitrous acid. *Water Pollution Control Federation*, **48**, 835–852.

Battistoni P., Paci B., Fatone F. and Pavan P. (2005). Phosphorus removal from supernatants at low concentration using packed and fluidized-bed reactors. *Industrial & Engineering Chemistry Research*, **44**, 6701–7.

Beier M. and Schneider Y. (2008). *Abschlussbericht Entwickelung von Bilanzmodellen für die Prozesse Deammonifikation und Nitritation zur Abbildung gross-and halb-technischer Anlagen* (Final report development of balance models for deammonification and nitritation processes to illustrate full and half technical scale installations). Leibniz University Hannover, Hannover: 39.

Blackburne R., Zhiguo Y. and Keller J. (2008). Partial nitrification to nitrite using low dissolved oxygen concentration as the main selection factor. *Biodegradation*, **19**(2), 303–312.

Carvalho C., Lemos P. C., Oehmen A. and Reis M. A. M. (2007). Denitrifying phosphorus removal: linking the process the process performance with the microbial community structure. *Water Research*, **41**, 4383–4396.

Cavinato C., Bolzonella D., Pavan P., Fatone F. and Cecchi F. (2013). Mesophilic and thermophilic anaerobic co-digestion of waste activated sludge and source sorted biowaste in pilot-and full-scale reactors. *Renewable Energy*, **55**, 260–265.

Cervantes F. J. (2009). Environmental Technologies to Treat Nitrogen Pollution. IWA Publishing, London.

Dockhorn T. (2009). About the Economy of Phosphorus Recovery. International Conference on Nutrient Recovery from Wastewater Streams.

Elser J. and Bennett E. (2011). Phosphorus cycle: a broken biogeochemical cycle. *Nature*, **478**, 29–31.

Fatone F., Dante M., Nota E., Di Fabio S., Frison N., Pavan P. and Fatone F. (2011). Biological short-cut nitrogen removal from anaerobic digestate in a demonstration sequencing batch reactor. *Chemical Engineering Transactions*, **24**, 1135–1140.

Frison N., Lampis S., Bolzonella D., Pavan P. and Fatone F. (2012). Two-stage start-up to achieve the stable via-nitrite pathway in a demonstration SBR for anaerobic codigestate treatment. *Industrial & Engineering Chemistry Research*, **51**(47), 15423–15430.

Frison N., Di Fabio S., Cavinato C., Pavan P. and Fatone F. (2013). Best available carbon sources to enhance the via-nitrite biological nutrients removal from supernatants of anaerobic co-digestion. *Chemical Engineering Journal*, **215**, 15–22.

Fux C., Velten S., Carozzi V., Solley D. and Keller J. (2006). Efficient and stable nitritation and denitritation of ammonium-rich sludge dewatering liquor using an SBR with continuous loading. *Water Research*, **40**(14), 2765–2775.

Ganigué R., López H., Balaguer M. D. and Colprim J. (2007). Partial ammonium oxidation to nitrite of high ammonium content urban landfill leachates. *Water Research*, **41**(15), 3317–3326.

Ganigué R., Volcke E. I., Puig S., Balaguer M. D. and Colprim J. (2012). "Impact of influent characteristics on a partial nitritation SBR treating high nitrogen loaded wastewater. *Bioresource Technology*, **111**, 62–69.

Gilbert N. (2009). The disappearing nutrient. *Nature*, **461**(8), 716–718.

Gu S. B., Wang S. Y., Yang Q., Yang P. and Peng Y. Z. (2012). Start up partial nitrification at low temperature with a real-time control strategy based on blower frequency and pH. *Bioresource Technology*, **112**, 34–41.

Guo J. H., Peng Y. Z., Wang S. Y., Zheng Y. N., Huang H. J. and Ge S. J. (2009). Effective and robust partial nitrification to nitrite by real-time aeration duration control in an SBR treating domestic wastewater. *Process Biochemistry*, **44**, 979–985.

Gustavsson D. J. I. (2010). Biological sludge liquor treatment at municipal wastewater treatment plants – a review, *Vatten*, **66**, 179–192.

Gustavsson D. J. I. and Jansen J. la Cour (2011). Dynamics of nitrogen oxides emission from a full-scale sludge liquor treatment plant with nitritation. *Water Science and Technology*, **63**(12), 2838–2845.

Hao X., Wang C., van Loosdrecht M. C. M. and Hu Y. (2013). Looking Beyond Struvite for P-Recovery. *Environmental Science & Technology*, **47**, 4965–4966.

Hellinga C., Schellen A. A. J. C., Mulder J. W., van Loosdrecht M. C. M. and Heijnen J. J. (1998). The SHARON_ process: an innovative method for nitrogen removal from ammonium-rich waste water. *Water Science and Technology*, **37**, 135–142.

Ivanov V., Kuang S., Stabnikov V. and Guo C. (2009). The removal of phosphorus from reject water in a municipal wastewater treatment plant using iron ore. *Journal of Chemical Technology and Biotechnology*, **84**, 78–82.

Jardin N., Thöle D. and Wett B. (2006). Treatment of sludge return liquors: experiences from the operation of full-scale plants. In: Proceedings of the Water Environment Federation, WEFTEC 2006, Dallas, 21–25 October, 5237–5255.

Joss A., Salzgeber D., Eugster J., König R., Rottermann K., Burger S., Fabijan P., Leumann S., Mohn J. and Siegrist H. (2009). Full-scale nitrogen removal from digester liquid with partial nitritation and anammox in one SBR. *Environmental Science & Technology*, **43**(14), 5301–5306.

Kim J., Park C., Kim T.-H., Lee M., Kim S., Kim S. W. and Lee J. (2003). Effects of various pretreatments for enhanced anaerobic digestion with waste activated sludge. *Journal of Bioscience and Bioengineering*, **95**(3), 271–275.

Kishida N., Kim J., Tsuneda S. and Sudo R. (2006). Anaerobic/oxic/anoxic granular sludge process as an effective nutrient removal process utilizing denitrifying polyphosphate-accumulating organisms. *Water Research*, **40**(12), 2303–2310.

Lee D. S., Che O. J. and Jong M. P. (2001). Biological nitrogen removal with enhanced phosphate uptake in a sequencing batch reactor using single sludge system. *Water Research*, **35**(16), 3968–3976.

Li X., Chen H., Hu L. F., Yu L., Chen Y. G. and Gu G. W. (2011). Pilot-scale waste activated sludge alkaline fermentation, fermentation liquid separation, and application of fermentation liquid to improve biological nutrient removal. *Environmental Science & Technology*, **45**, 1834–1839.

Longo S., Katsou E., Malamis S., Frison N., Renzi D. and Fatone F. (2015). Recovery of volatile fatty acids from fermentation of sewage sludge in municipal wastewater treatment plants. *Bioresource Technology*, **175**, 436–444.

Malamis S., Katsou E., Di Fabio S., Bolzonella D. and Fatone F. (2013). Biological nutrients removal from the supernatant originating from the anaerobic digestion of the organic fraction of municipal solid waste. *CRC Critical Reviews in Biotechnology*, **34**(3), 244–257.

Morse G. K., Brett S. W., Guy J. A. and Lester J. N. (1998). Review: Phosphorus removal and recovery technologies. *Science of The Total Environment*, **212**, 69–81.

Mulder A. (2003). The quest for sustainable nitrogen removal technologies. *Water Science and Technology*, **48**(1), 67–75.

Münch, Elisabeth V. and Keith Barr. (2001). Controlled struvite crystallisation for removing phosphorus from anaerobic digester side streams. *Water Research*, **35**(1), 151–159.

Oleszkiewicz J. A. and Barnard J. L. (2006). Nutrient removal technology in North America and the European Union: a review. *Water Quality Research Journal of Canada*, **41**, 449–462.

Peng Y. and Zhu G. (2006). Biological nitrogen removal with nitrification and denitrification via nitrite pathway. *Applied Microbiology and Biotechnology*, **73**, 15–26.

Peng Y.-Z., Wu C.-Y., Wang R.-D. and Li X.-L. (2011). Denitrifying phosphorus removal with nitrite by a real-time step feed sequencing batch reactor. *Journal of Chemical Technology and Biotechnology*, **86**, 541–546.

Pitman A. R. (1999). Management of biological nutrient removal plant sludges – change the paradigms? *Water Research*, **33**, 1141–1146.

Pollice A., Laera G., Saturno D. and Giordano C. (2008). Effects of sludge retention time on the performance of a membrane bioreactor treating municipal sewage. *Journal of Membrane Science*, **317**(1), 65–70.

Seongjun P., Bae W. and Rittmann B. E. (2010). Multi-species nitrifying biofilm model (MSNBM) including free ammonia and free nitrous acid inhibition and oxygen limitation. *Biotechnology and Bioengineering*, **105**(6), 1115–1130.

STOWA (2012). Explorative research on innovative nitrogen recovery, Rapport 51.

Su G., Huo M., Yuan Z., Wang S. and Peng Y. (2013). Hydrolysis, acidification and dewaterability of waste activated sludge under alkaline conditions: combined effects of NaOH and Ca(OH) 2. *Bioresource Technology*, **136**, 237–243.

Taruya T., Ueno Y. and Fujii M. (2000). Development of phosphorus resource recycling process from sewage, Poster Paper, First World Water Congress of the IWA, Paris, 3–7 July.

Tong J. and Chen Y. (2007). Enhanced biological phosphorus removal driven by short-chain fatty acids produced from waste activated sludge alkaline fermentation. *Environmental Science & Technology*, **41**, 7126–7130.

van Dongen U., Jetten M. S. M. and van Loosdrecht M. C. M. (2001). The Sharon – Anammox process for treatment of ammonium rich wastewater. *Water Science and Technology*, **44**(1), 153–160.

Volcke E. I. P., van Loosdrecht M. C. M. and Vanrolleghem P. A. (2007). Interaction between control and design of a SHARON reactor: economic considerations in a plant-wide (BSM2) context. *Water Science and Technology*, **56**(9), 117–125.

Wett B. (2007). Development and implementation of a robust deammonification process. *Water Science and Technology*, **56**(7), 81–88.

Wu H., Gao J., Yang D., Zhou Q. and Liu W. (2010). Alkaline fermentation of primary sludge for short-chain fatty acids accumulation and mechanism. *Chemical Engineering Journal*, **160**, 1–7.

Ye L., Tang B., Zhao K. F., Pijuan M. and Peng Y. Z. (2009). Nitrogen removal via nitrite in domestic wastewater treatment using combined salt inhibition and on-line process control. *Water Science and Technology*, **60**(6), 1633–1639.

Yuan H., Chen Y., Zhang H., Jiang S., Zhou Q. and Gu G. (2006). Improved bioproduction of short-chain fatty acids (scfas) from excess sludge under alkaline conditions. *Environmental Science & Technology*, **40**, 2025–2029.

Zhang S., Peng Y., Wang S., Zheng S. and Gou S. (2007). Organic matter and concentrated nitrogen removal by shortcut nitrification and denitrification from mature municipal landfill leachate. *Journal of Environmental Sciences*, **19**, 647–651.

Zheng X., Chen Y. and Liu C. (2010). Waste activated sludge alkaline fermentation liquid as carbon source for biological nutrients removal in anaerobic followed by alternating aerobic–anoxic sequneching batch reactors. *Chinese Journal of Chemical Engineering*, **18**, 478–485.

Zhu X. Y. and Chen Y.G. (2011). Reduction of N2O and NO generation in anaerobic-aerobic biological wastewater treatment process by using sludge alkaline fermentation liquid. *Environmental Science & Technology*, **45**, 2137–2145.