ACS13221 v2

ANNEXES
WASTEWATER TO ENERGY

A Technical Note for Utility Managers and Decision Makers on Urban Sanitation in East Asian Countries

January 2015

WORLD BANK GROUP
Water

WATER PARTNERSHIP PROGRAM

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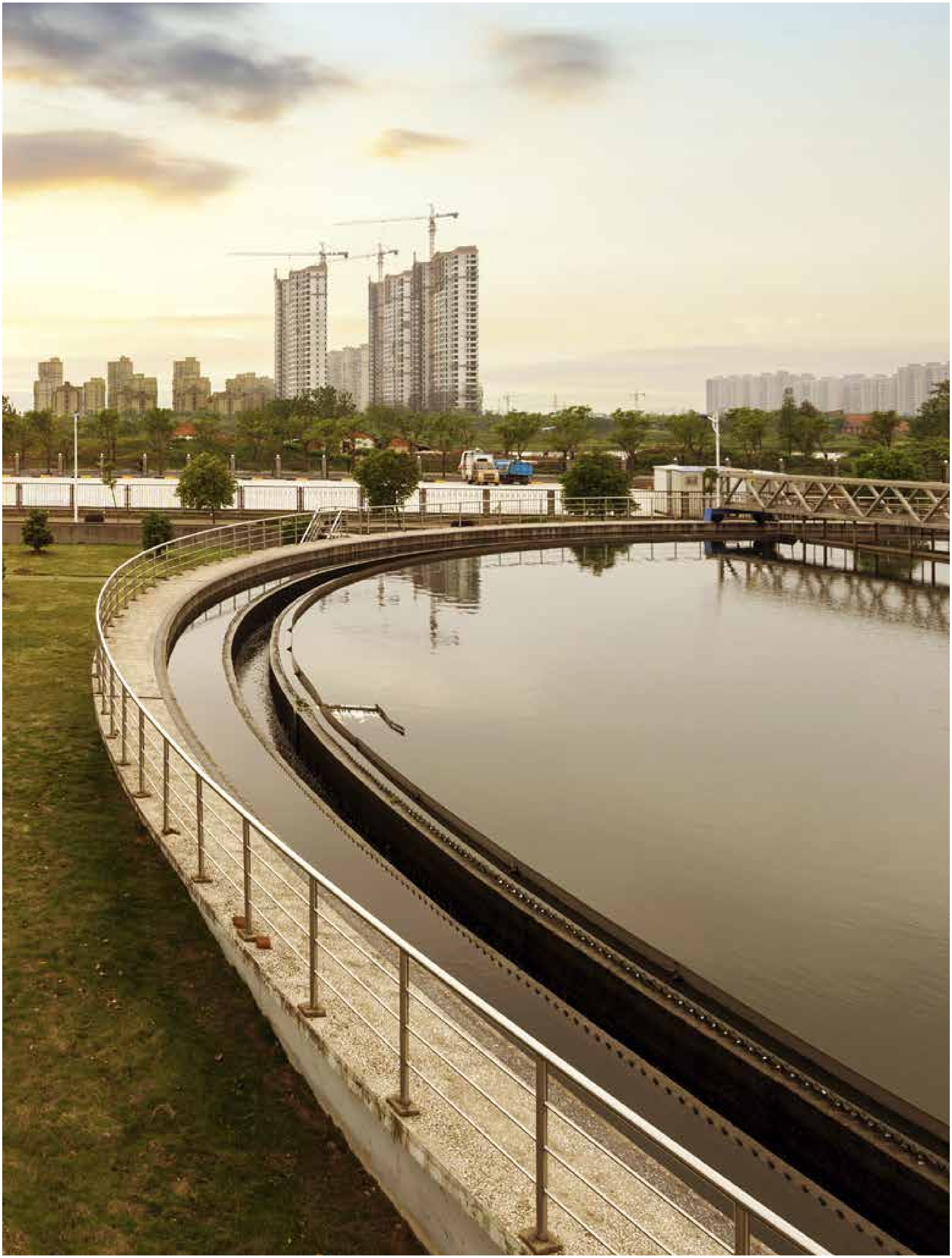
This document provides complementary information to the main report. In particular, annex 3 includes the detailed analysis and description of the case studies, which are summarized in section 2 of the main report.



ANNEX 1: ENERGY EFFICIENCY EFFORTS IN EUROPE AND UNITED STATES

Efforts to Optimize Energy Consumption in Europe

The activities related to energy optimization (frequently known as "energy efficiency") at wastewater treatment plants (WWTPs) in Europe started about fifteen years ago and primarily focused on conventional activated sludge (CAS). Energy optimization entails two components: (a) minimization of energy consumption and (b) maximization of biogas production, and they include the following two methods:

- Benchmarking
- Energy manuals

The former provides up-to-date information on what is technically and financially feasible, and the latter provides practical advice on how to implement energy-efficient designs and rehabilitate facilities to improve their energy balance.

Benchmarking

Benchmarking not only provides practical knowledge about what is technically and financially feasible, but it also encourages competition among and within WWTP utilities and leads to generally better performance and reduced cost for participants over time. The best known European benchmarking projects are the following (in chronological order, indicating the year of project start):

- Norway — 1998: NORVAR (Norsk Vann BA) (WERF 2010a)
- Netherlands — 1999: Dutch Benchmarking Project (WERF 2010a)
- Austria — 1999: Benchmarking in Sanitary Engineering (BLFUW 2001)
- Denmark — 2000: DANVA Benchmarking Project (WERF 2010a)
- Finland — 2003: Finnish Benchmarking Project (WERF 2010a)
- Germany — 2005: Rhineland-Palatinate (MUFV-RP 2006)
  2006: Bavaria (Graf 2008)
  2006: Baden-Württemberg (DWA-BW 2010)

Most benchmarking projects focus on two aspects:

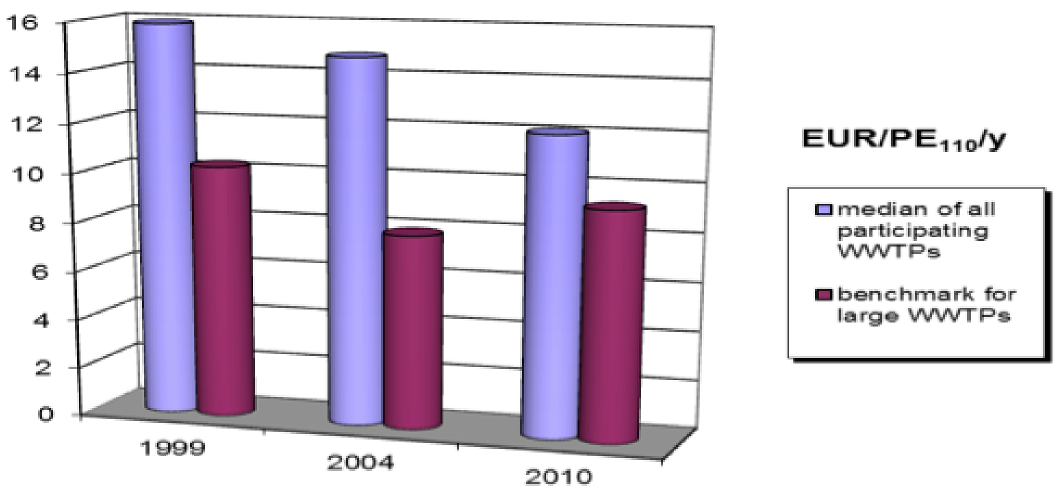
- Performance-related indicators, such as effluent quality, removal efficiencies, electricity consumption, and so on
- Cost-related indicators, typically both pertaining to CAPEX and OPEX

Generally, benchmarking results are difficult to transfer from one region to another. Local conditions, such as specific climate, specific discharge requirements, and specific unit costs often have quite different overall impacts in different regions and countries. This needs to be considered when applying "lessons learned" from one region to another.

Most of the above benchmarking projects have been repeated ever since their start, typically annually or every two years, so there is currently good and significant knowledge about the sanitation sector in many different European regions, and it is interesting to see the development of project indicators over time. Gradual OPEX reductions, as depicted in figure A1-1, are to a large extent connected to energy optimization efforts.

Detailed benchmarking reports are not usually public documents.

Figure A1-1: Development of OPEX for large Austrian WWTPs participating in benchmarking



Sources: Wett et al. 2007; Lindtner et al. 2012.
Note: EUR/PE$_{110}$/y x 16.67 = EUR/kg BOD$_5$/y.

Energy manuals

Energy manuals have an even longer history in Europe than benchmarking projects. The first manuals were developed in Switzerland in 1994, followed by various manuals in Germany, trying to take specific conditions and technological developments into account. The 1994 Swiss Energy Manual for WWTPs was updated most recently in 2010 and can now be considered as representing the state of the art of knowledge on this subject. The European energy manuals that are worth mentioning are the following (in chronological order, indicating the year issued):

- Switzerland 1994 (BUWAL 1994)
- Germany 1999 North Rhine Westphalia (MURL 1999)

  2008 German Association for Water,

  Wastewater, and Waste (DWA-BW 2008)

  2008 Umweltbundesamt (UBA 2008)
- Switzerland 2010 (VSA 2010)

Energy manuals usually include two main components:

- Technical, energy-related guidebooks for WWTPs, providing background information on energy requirements at WWTPs and on energy characteristics of all kinds of electro-mechanical equipment
- Practical recommendations and strategies for implementing energy optimization at WWTPs, covering minimization of energy consumption and maximization of biogas production and utilization

The main drawbacks of these manuals are that they are (a) mostly written in German, which hampers international application, and (b) focused primarily on conventional activated sludge (CAS) systems under European climate conditions, and therefore not as useful for other process technologies, such as upflow anaerobic sludge blanket (UASB), trickling filter, and waste stabilization pond (WSP) technologies. This does not mean some of the described subprocesses—for instance, sludge dewatering—could not also be applied together with technologies other than CAS.

Energy optimization

A typical energy optimization process for WWTPs has two stages:

- Energy screening, in which an existing WWTP's energy condition is checked by using seven indicators (see table A1-1). For each of these, a "target value" and an "ideal value" are defined. If any of the seven parameters fails to meet one or even several target values, a detailed analysis is recommended.
- Detailed energy analysis, which is based on the following:
  - Individual analysis of all treatment stages and all electro-mechanical installations
  - Definition of (a) short-term measures and (b) medium- and long-term measures
  - Evaluation of financial viability of recommended measures

Müller and others (2004) evaluated the energy analysis of 344 WWTPs in North Rhine Westphalia, Germany, and produced several major findings:

- Energy cost can be reduced by an average of 50 percent.
- Extrapolation of the findings in North Rhine Westphalia indicates a savings potential in Germany equal to US$4 billion–5 billion (EUR3 billion–4 billion) over a fifteen-year period.
- Energy optimization is usually financially attractive for WWTPs—that is, the NPV (net present value) of savings and investments is positive.

For Switzerland, where more than two-thirds of all WWTPs have already undergone energy analysis, Müller and others (2006) reported similar practical results:

- Energy cost at optimized WWTPs had been reduced by an average of 38 percent.
- 33 percent of the cost reduction was due to improved efficiency; 67 percent was due to increased energy production from biogas.
- Major efficiency increases were realized in the biological stage and with improved energy management.

Success stories

Recent reports demonstrate WWTPs are capable of reaching a positive energy balance—that is, some wastewater facilities do produce more energy than they consume. The first example was reported by Wett and others (2007), who described the Strass WWTP in Austria, a 167,000 PE_{60} CAS plant. An 8 percent electricity surplus was achieved, even though the plant was practicing enhanced nutrient removal. These results had been reached by 2005 through a combination of several efficiency-focused measures, and they sparked widespread interest. This facility is used as a benchmark in Singapore, among other countries (Cao 2011). In the United States it has been included as a case study in an analysis by the Water Environment Research Foundation (WERF 2010a); and Kang and others (2010) have developed a "four steps to energy self-sufficiency" road map for U.S. WWTPs.

Two more European facilities provide additional examples of plants that have achieved positive energy balances without co-digestion of external waste: Keil (2013) reports on the Bad Ischl WWTP in Austria, a 100,000 PE_{60} plant that has produced an annual electricity surplus between 2 and 12 percent since 2009. Sandino and others (2013) report on Odense

WWTP, Denmark, a 385,000 PE_{60} plant that has had similar results.

Recently, a strong trend has been observed toward co-digestion of organic waste, which facilitates the achievement of energy independence. Case study 5 in this report describes a WWTP that achieved 100 percent electricity coverage through co-digestion. Other successful examples are also cited.

These results demonstrate that energy independence of CAS systems is a feasible target almost anywhere if co-digestion is applied. Without co-digestion, energy independence of CAS systems is feasible, but it is much more difficult to achieve. The successful examples make clear that accomplishing it without co-digestion requires know-how, a high degree of automation, skilled operators, a clear target, and a commitment to its implementation.

Table A1-1: State of the art indicators for energy screening of CAS, based on European experiences with WWTP benchmarking and energy optimization to date

TARGET VALUES and IDEAL VALUES for energy screening of WWTPs												
PARAMETER			ACTUAL WWTP INFLUENT LOAD (PE)									
			2,000 - 5,000		5,000 - 10,000		10,000 - 30,000		30,000 - 100,000		> 100,000	
			Target	Ideal	Target	Ideal	Target	Ideal	Target	Ideal	Target	Ideal
E_{total}	**Total electricity consumption per PE * / ** / ******											
	C (sludge age < 6 days), including anaerobic sludge digester	kWh/PE/y	---	---	30	23	(27)	(21)	(24)	(18)	---	---
	C+N (sludge age ≈ 13 days), including anaerobic sludge digester	kWh/PE/y	---	---	39	30	34	26	30	23	26	20
	C+N (sludge age > 25 days), with extended aeration	kWh/PE/y	54	41	46	35	40	31	---	---	---	---
E_{bio}	**Electricity consumption in biological stage per PE *** / ******											
	C (sludge age < 6 days), including anaerobic sludge digester	kWh/PE/y	---	---	20	15	(18)	(14)	(17)	(13)	---	---
	C+N (sludge age ≈ 13 days), including anaerobic sludge digester	kWh/PE/y	---	---	29	22	25	19	23	18	21	16
	C+N (sludge age > 25 days), with extended aeration	kWh/PE/y	41	32	36	28	31	24	---	---	---	---
N_1	**Percentage of biogas utilization**	%	---	---	95%	97%	97%	98%	98%	99%	98%	99%
N_2	**Percentage of biogas used for electricity generation**	%	---	---	27%	29%	31%	33%	33%	35%	35%	37%
N_3	**Specific biogas generation yield**											
	C	L / kg VSS	---	---	500	525	(500)	(525)	(500)	(525)	---	---
	C+N	L / kg VSS	---	---	450	475	450	475	450	475	450	475
S_e	**Supply of total electricity requirements from own biogas utilization * / ** / ******											
	C	%	---	---	50%	67%	(64%)	(86%)	(74%)	(97%)	---	---
	C+N	%	---	---	39%	52%	52%	69%	60%	80%	70%	92%
S_t	**Supply of total thermal heat requirements from own biogas utilization**	%	---	---	90%	95%	95%	97%	97%	98%	98%	99%

* In case of pumping stations: E_{total}: Add + 0,5 kWh/PE/y for each m pumping head. S_e: Deduct accordingly.

** In case of filtration stage: E_{total}: Target + 3 kWh/PE/y, Ideal + 2 kWh/PE/y, (for WWTPs < 30,000 PE additional + 1 kWh/PE/y). S_e: Deduct accordingly.

*** The "biological stage" shall include energy consumption for: aeration / blowers, mixers, internal recirculation, recirculation via secondary sedimentation.

**** In case of process technology other than CAS: E_{total}: Biofilters target + 15 kWh/PE/y, Ideal + 10 kWh/PE/y, MBR (Membrane bioreactors) target + 25 kWh/PE/y, Ideal + 20 kWh/PE/y, Fixed Bed Reactors: as CAS. S_e: Deduct accordingly.

OBSERVATIONS:

Target: Value which can be achieved realistically, based on practical experiences.

Ideal: Value which can be achieved under optimum conditions.

PE: Actual annual average influent load, based on 1 PE = 120 g COD/PE/d

All values are derived for WWTPs with Primary Sedimentation, and normal municipal wastewater with normal C / N ratio.

Values in brackets hold true for C. Only of temporary relevance, since not able to meet effluent requirements for such WWTP size.

C: carbon removal (BOD_5, COD, TSS) as only treatment target.

C+N: carbon + nutrient removal (nitrogen, phosphorus) as treatment target.

Source: VSA 2010.

Note: kWh/PE/y x 16.67 = kWh/kg BOD_5/y.

Efforts to Optimize Energy Consumption in the United States

Energy optimization at WWTPs is receiving increased attention in the United States, motivated by continuously increasing energy unit costs and discussions about climate change and greenhouse gas (GHG) emissions.

WERF and the Water Environment Federation (WEF) have done considerable work in this area. WERF started off with a series of reports on best practices and case studies regarding energy efficiency at WWTPs (see, for example, WERF 2010a, 2010b, 2010c, and 2011a). WEF developed an energy road map (WEF 2012) (see figure A1-2) and subsequently specified this approach further (WEF 2013a). Generally, most large North American consulting companies have been involved in this process in one way or another. Many successful case studies make references to European or Asian facilities. Also, in recent years, an increased number of presentations at U.S. conferences and workshops have dealt with energy efficiency at WWTPs.

Figure A1-2: Main topics of the U.S. energy road map and levels of progression

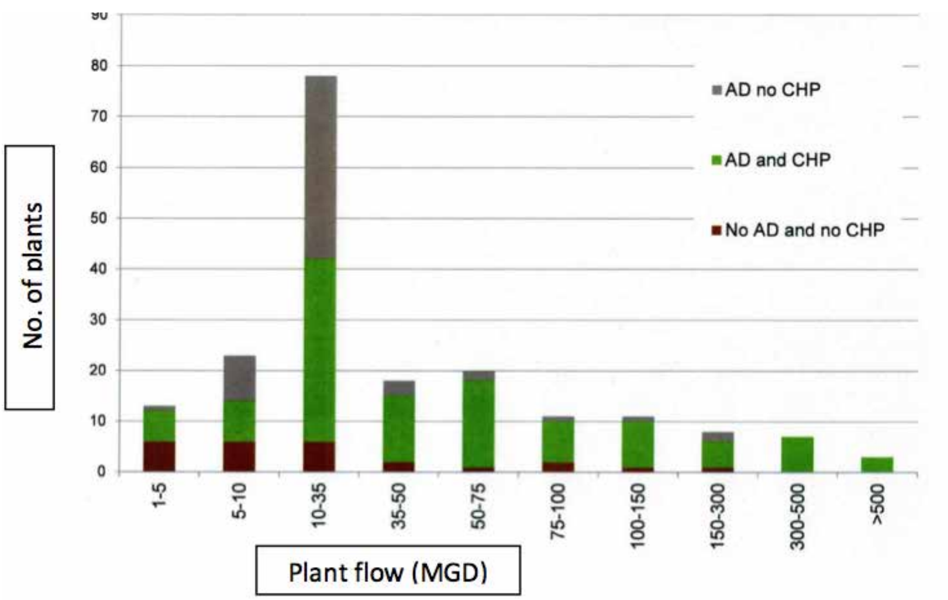
Source: WEF 2012.

ANNEX 2: SLUDGE DIGESTERS AND RENEWABLE ELECTRICITY POTENTIAL FROM BIOGAS IN THE UNITED STATES AND EUROPE

Sludge Digesters in the United States and Europe

According to Renewable Waste Intelligence (RWI 2013), 1,500 WWTPs (that is, less than 10 percent of the approximately 16,500 WWTPs in the United States) have anaerobic digesters, and only 250 of them use the biogas they produce (biogas from the other 1,250 is flared off). This information matches a U.S. Environmental Protection Agency (EPA) estimate cited by WERF (2012a) and Stone and Willis (2012), stating that less than 20 percent of large WWTPs with anaerobic digestion employ combined heat and power (CHP). The two studies present a survey on the use of anaerobic digesters and CHP that received over 200 responses from thirty-six states, Australia, and Canada. The respondents' rates of anaerobic digestion and CHP are presented in figure A2-1.

Figure A2-1: Rated WWTP flow (MGD) versus biogas use



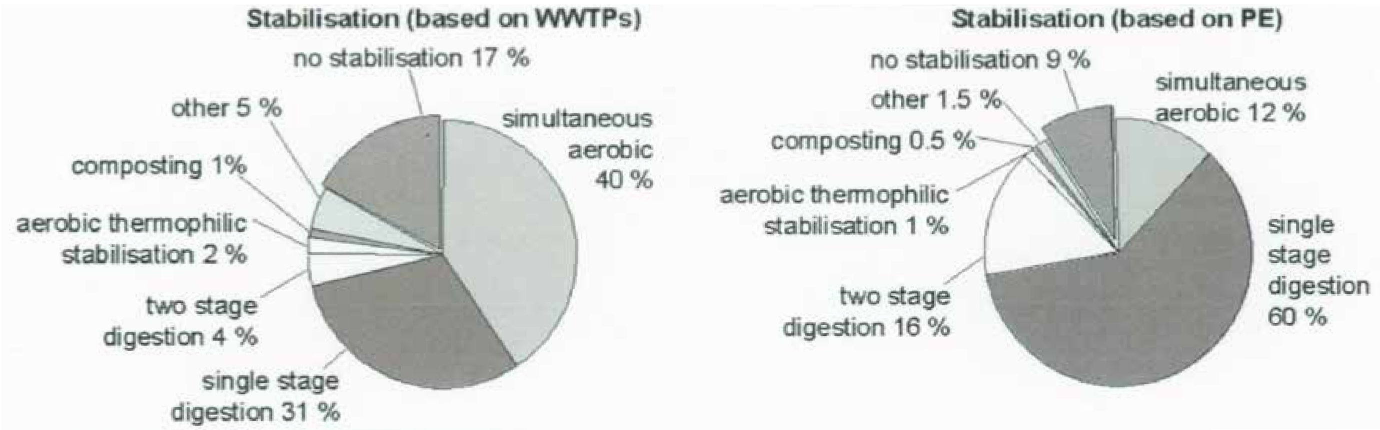
Source: WERF (2012a) and Stone and Willis (2012)

The figure shows that across the complete range of responding plant sizes from 1 million to more than 500 million U.S. gallons per day (MGD; approximately 10,000–7,000,000 PE_{60}), anaerobic digestion (AD) is being applied. Although larger plants are more likely to be equipped with AD, many of the smallest have it as well. Most plants with anaerobic digestion also have CHP. Only the intermediary group between 5 and 35 MGD has a relatively high number of plants without CHP.

This analysis demonstrates that even small plants down to 1 MGD (10,000 PE_{60}) can successfully argue a case for CHP in the United States.

In the case of Europe, Meda and others (2006) present the results of a survey for Germany. This survey covered about two-thirds of total organic loading capacity installed in Germany and about one-third of all WWTPs in absolute numbers. They found that 76 percent of total PE_{60} capacity feature anaerobic digestion, whereas 35 percent of WWTPs have it—that is, mainly medium and large WWTPs use anaerobic digestion (see figure A2-2).

Figure A2-2: Installed sludge stabilization technologies in Germany, based on number of WWTPs and on PE_{60}



Source : Meda et al (2006)

Renewable Electricity Potential from Biogas at WWTPs in the United States and Europe

According to WEF (2011), more than 500 plants in the United States currently use anaerobic digestion without CHP. If all of them were to install CHP systems, they could generate approximately 340 megawatts (MW) of electricity. The total CHP potential from all WWTPs in the United States over 1 MGD is quantified as 600 MW (Stone and Willis 2012). According to WERF (2012a), "WWTPs have the potential to generate an additional 200 to 400 MW of power from biogas."

Assuming the 600 MW capacity is in operation all year round, this would result in the production of about 5 million megawatt hours per year (MWh/y). Although this electricity production is substantial, it is only a little over 10 percent of the actual consumption of all U.S. WWTPs of about 40 million MWh/y (WERF 2010b).

In Germany, with 1 PE_{60} able to produce an average of 15 kWh/PE_{60}/y, the total electricity production potential from sludge equals about 1.8 million MWh/y at present. That is equivalent to more than 40 percent of the actual electricity consumption of all WWTPs of 4.2 million MWh/y (DWA 2013b). As mentioned in the case of the United States, to further increase that percentage, utilities would have to combine the effects of reduced electricity consumption at WWTPs with measures to further increase biogas quantities.

ANNEX 3: CASE STUDIES

CASE STUDY
1: CAS + SLUDGE DIGESTION

1.1. BACKGROUND, PROCESS DESCRIPTION

1.1.1. Data sources

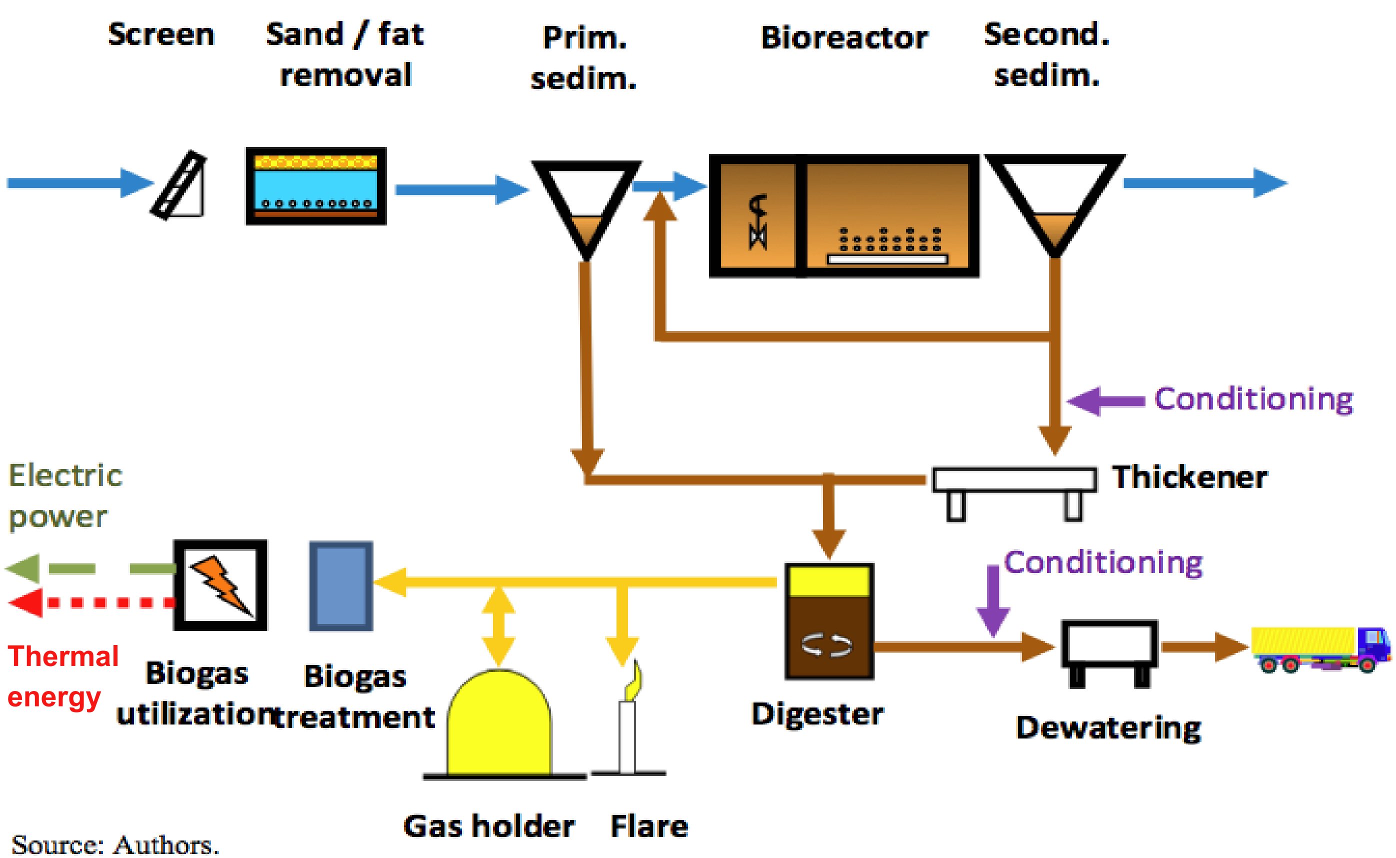
Case study 1 is based on publicly accessible data from Europe, which are based on thousands of WWTPs, so that sources such as benchmarking results for CAS and national/regional wastewater treatment statistics represent a solid data base. The core data for case study 1 were taken from Germany and Austria, which have similar treatment requirements and technology standards.

1.1.2. Wastewater management

The analysis made in this case study assumes the reader is familiar with the conventional activated sludge (CAS) process. All CAS technology variations have in common that they produce waste activated sludge. Extended aeration is excluded from in-depth analysis in case study 1, since extended aeration does sludge stabilization inside the aeration tanks and does not produce biogas for energy generation.

The typical CAS technology used in this case study (see figure CS1-1) consists of preliminary treatment (screen, sand/fat removal), followed by primary sedimentation tank (PST) and, subsequently, aeration tank (AT) and secondary sedimentation tank (SST).

Figure CS1-1: Simplified flow scheme of CAS and mesophilic sludge digestion



Source: Authors.

1.1.3. Sludge management

The assumption in the following analysis is that all sludge produced in wastewater treatment, both primary and secondary, will be thickened and subsequently anaerobically digested. Thickening is important, since it reduces sludge quantities considerably and thus allows for smaller digester designs, which produces CAPEX savings. Primary sludge thickens well and rapidly by gravity in the PST, up to some 3 percent dry solids (DS). Therefore, separate thickeners for primary sludge are frequently not necessary. Since waste activated sludge rarely exceeds 1 percent DS, it requires separate thickening prior to digestion. To maximize thickening efficiency and thus minimize digester cost, waste activated sludge is usually thickened mechanically to about 6 percent DS. Anaerobically digested sludge is generally dewatered prior to reuse or disposal.

See table CS1-4 for general design parameters and key characteristics of anaerobic digesters.

1.1.4. Energy management

The biogas produced in the digesters can be utilized for the production of electric and/or thermal energy, in their combined form also called combined heat and power (CHP) production. Prior to its utilization, the gas is balanced in a gas holder and purified, as required. A flare is typically used to burn gas that cannot be utilized and for emergency situations.

See table CS1-5 for general design parameters and key characteristics of biogas systems.

1.2. ANALYSIS

1.2.1. Wastewater influent and effluent

Most benchmarking initiatives assume that participating WWTPs comply with legal effluent criteria and therefore do not present specific information on influent and effluent wastewater characteristics. The national wastewater associations of Germany and Austria—DWA and OEWAV, respectively—compile these data in an annual WWTP performance analysis. Table CS1-1 provides data for the year 2012 (DWA 2013a).

Table CS1-1: Average influent and effluent data of WWTPs in Germany and Austria in 2012

			Germany	Austria
Number of WWTPs			5,917	906
WASTEWATER QUANTITY				
Specific wastewater production		$m^3/PE_{60}/y$	80	67
		$L/PE_{60}/d$	219	184
WASTEWATER QUALITY				
COD	Influent	mg/L	548	656
	Effluent	mg/L	27	44
	Elimination	%	95	93
BOD_5*	Influent	mg/L	255	n.a.
	Effluent	mg/L	4.5	n.a.
	Elimination	%	98	n.a.
N_{total}	Influent	mg/L	51	43
	Effluent	mg/L	9	9
	Elimination	%	82	79
NH_4-N	Effluent	mg/L	1.2	1.2
NO_3-N	Effluent	mg/L	6.0	5.7
P_{total}	Influent	mg/L	7.9	7.5
	Effluent	mg/L	0.7	0.7
	Elimination	%	91	91

Source: DWA 2013a.

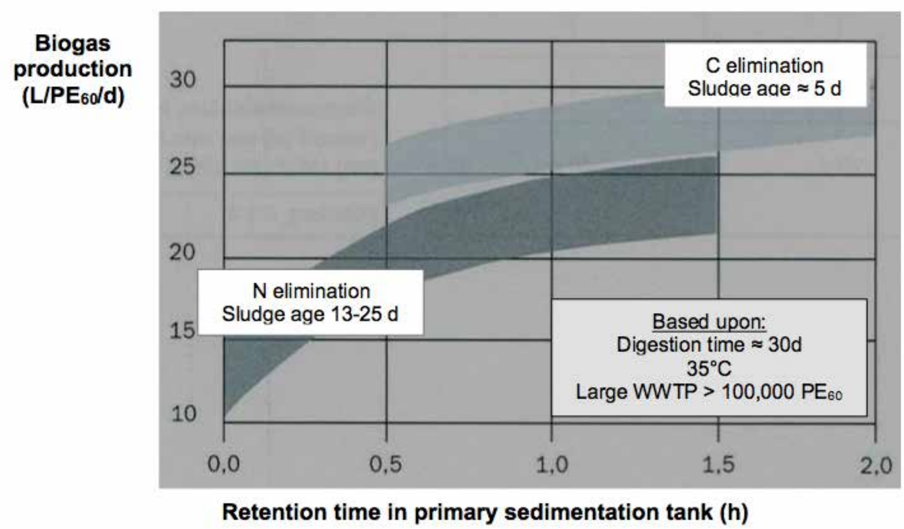
*Note: 1 cap = 60 g BOD_5/d in Germany and Austria; 1 PE_{60} = 1.0 cap in Germany and Austria; $m^3/PE_{60}/y$ = m^3/cap/y in Germany and Austria; $L/PE_{60}/d$ = L/cap/d in Germany and Austria. *Data from 2010. n.a. = not available.*

With a chemical oxygen demand (COD) of about 600 mg/L, the raw wastewater is of average strength. This allows for efficient application of primary sedimentation tanks (PSTs), which is advantageous for high biogas production rates. On the other hand, nitrogen reduction is high at these WWTPs; hence, on top of high energy requirements for CAS, in general, there is additional energy consumption for the oxidation of nitrogen. Under these nutrient reduction conditions, WWTPs are usually not able to recover all 100 percent of their energy needs from the biogas.

1.2.2. Biogas production and potential for energy generation

Project-specific biogas production rates can be calculated with the design values provided in table CS1-5. However, this requires the prior definition of sludge quantities and qualities. If these definitions are not available, the summaries in figure CS1-2 of biogas production for different conditions can be used for a rough first estimate. The indicated ranges are a reliable reflection of what is actually observed in practice, too.

Figure CS1-2: Daily biogas production rate per PE_{60} at large WWTPs



Source: VSA 2010.
Note: L/PE_{60}/d x 16.67 = L/kg BOD_5/d.

WWTPs with the best treatment performance (enhanced nitrogen (N) removal) show the lowest biogas production rates. This is due to short–retention time primary settling tanks (PSTs), usually required for efficient denitrification, and to high sludge age in the bioreactors, which leads to increased consumption of carbon, reduced quantities of fresh primary sludge, and partially stabilized secondary sludge. A typical WWTP under these conditions might produce just about 20 L/PE_{60}/d.

Whereas WWTPs that only provide for carbon (C) removal and include large PSTs can reach higher biogas production rates of about 25–30 L/PE_{60}/d, the decisive question for EAP and other warm countries will be in which loading range their CAS plants operate. Nitrification in warm climates is much faster than in cold climates. Hence, even if a WWTP is designed for carbon removal only, it will nitrify its wastewaters unless due attention is paid to operating it in a loading range that does not yet lead to nitrification.

Note that the above figures hold true for WWTPs larger than 100,000 PE_{60} and are based on mesophilic digesters with a retention time of thirty days. For other conditions, consider the following:

- In cases of shorter retention times in the digester of just about fifteen to twenty days, the values of figure CS1-2 should be reduced by 10 percent.
- Likewise, small and medium WWTPs will produce somewhat reduced biogas quantities, mainly due to reduced efficiencies caused by stronger impacts of peak loading periods. It is possible to assume 5 percent lower values for a 50,000 PE_{60} plant and 10 percent lower for a 20,000 PE_{60} plant.

Methane's calorific value of 10 kWh/m^3 is important to estimating the potential for energy generation from biogas production. Since biogas contains about 60 to 70 percent methane, its calorific value usually ranges from 6.0 to 7.0 kWh/m^3 after cleaning and moisture removal. The most common and economical options for utilizing this resource are co-generation and microturbines. Consequently, the overwhelming majority of the WWTPs considered for case study 1 also use one or the other of these technologies.

Based on the above described biogas production rates and typical electric efficiencies for CHP, as indicated in figure CS1-31, the potential for energy generation through CAS + mesophilic digestion is as shown in table CS1-2.

Table CS1-2: Biogas and power generation potential of CAS + mesophilic digester at large WWTPs > 100,000 PE_{60}

	Retention time in PST (h)			
	0.5	1.0	1.5	2.0
Biogas production				
- N elimination (L/PE_{60}/d)	**20**	**22**	**23**	**—**
- C elimination (L/PE_{60}/d)	**24**	**27**	**28**	**29**
Electric efficiency CHP (%)	33	33	33	33
Thermal efficiency CHP (%)	50	50	50	50
Calorific value of biogas (kWh/m^3)	6.3	6.3	6.3	6.3
Power generation				
- N elimination (kWh/PE_{60}/year)	**15.2**	**16.7**	**17.5**	**—**
- C elimination (kWh/PE_{60}/year)	**18.2**	**20.5**	**21.2**	**22.0**
Thermal energy generation				
- N elimination (kWh/PE_{60}/year)	23.0	25.3	26.4	—
- C elimination (kWh/PE_{60}/year)	27.6	31.0	32.2	33.3
Total energy generation				
- N elimination (kWh/PE_{60}/year)	38.2	42.0	43.9	—
- C elimination (kWh/PE_{60}/year)	45.8	51.5	53.4	55.3

Source: Authors' calculation.

Note: L/PE_{60}/d x 16.67 = L/kg BOD_5/d; kWh/PE_{60}/y x 16.67 = kWh/kg BOD_5/y.

Generally, the calculated values describe average conditions. For specific conditions, a certain amount of adjustment can be necessary—for instance, in the case of very large co-generation units with higher than assumed electric efficiency or of microturbines or of a different calorific value of the biogas. Typically, those variations can affect the cited energy generation values by not more than about +/– 10 percent.

These values in table CS1-2 are confirmed by actual benchmarking data from Austria (Lindtner 2011) and Germany (DWA-BW 2010; Graf 2010). These practice data also show the electricity potential is not fully exploited at present. Of the biogas in the region, 20–30 percent still goes unused or is just utilized in burners for heat production. Notwithstanding, there is a strong trend toward further increasing electric power generation at WWTPs, particularly since energy unit cost is rising fast in Central Europe.

1.2.3. Operation capacity needs, biogas safety

The key operational issues concerning anaerobic digesters involve specific operational problems and safety concerns related to the management of explosive biogas. The most relevant aspects are those discussed below:

- Safety concerns
- Digester foaming
- Deposits in the digester
- Insufficient biogas treatment

Safety concerns

Although accidents occur, they are very rare. Biogas operating conditions have a risk of explosion only when 5–15 percent biogas is mixed with 85–95 percent air. It is important to prevent large quantities of air from entering biogas systems. Micro-aeration of digesters, used for H_2S removal, does not present a safety problem. Confined areas can sometimes pose risks for workers if methane and carbon monoxide accumulate.

The Austrian wastewater association published a compilation of all accidents related to biogas explosions in the previous twenty years (OEWAV 2006). The main recommendations that can be drawn from those (few) accidents are the following:

- When repairing a digester/gas holder/sludge pipe, instruct workers very clearly about the necessary safety precautions. Particularly avoid any sources of spark ignition, such as welding, metal work, smoking, and so on when this ignition spark might get into contact with biogas.
- Always assign one person who is clearly in charge for supervision of work and safety precautions.
- At design, implementation, and operation stages, always adhere to standard safety precautions. Important precautions include those mentioned by Germany's wastewater association (DWA 1996):
    - Maintain a slight overpressure in biogas systems to avoid entry of air.
    - Design sludge feeding and abstraction based on the sludge replacement principle: the sludge entering the digester should push the same amount of sludge out of the digester. This will ensure the overpressure in the biogas systems can be maintained.
    - Take particular care when removing scum from the digester surface. This is an action during which considerable amounts of biogas could be released to the open air.

- Separate digester, gas holder, and biogas utilization through appropriate valves. Note that water-filled foam traps are safer than mechanical valves, which tend not to be 100 percent foolproof.
- Construct gas pipes and gas installations from corrosion-resistant material only— for instance, grade 1.4571 stainless steel or high-density polyethylene (HDPE). Take temperature effects on HDPE into due account.
- Condensate removal must work automatically. Manual condensate removal installations prove unreliable in practice.
- Favor pipe welding over pipe flanges. Also, compensators should preferably be welded to the pipes.
- The ventilation of adjacent rooms and pipe collectors should preferably be done through slight overpressure inside these rooms, rather than through air suction out of them.
- Install and regularly maintain gas warning installations, particularly in those locations where biogas accumulates. These installations are supposed to trigger an alarm before critical concentrations are achieved.
- For any repair and maintenance work, use special tools that cannot cause ignition sparks.
- Respect special operation and maintenance instructions from the providers of biogas and digester equipment.
- Respect local legal requirements for biogas systems.

Digester foaming

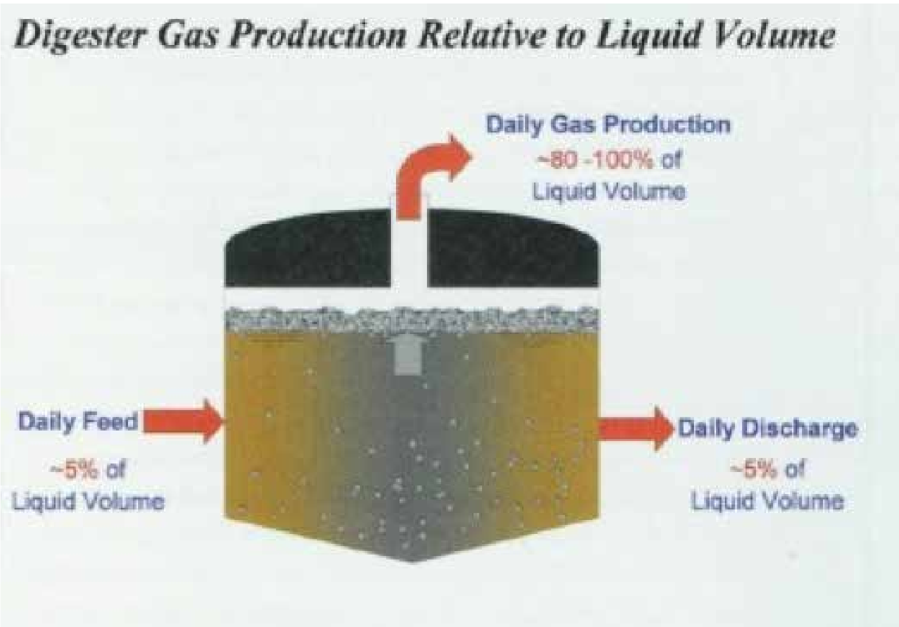
Digester foaming is occasionally observed in digesters at WWTPs. Once it appears, it can last from one day to several weeks. Persistent foaming is rare. The practical impact of foaming is that the foam tends to clog the biogas system, and it can affect pumps and mixing/heating devices. As a consequence, foaming results in extra manual repair and cleaning work. Also, persistent foaming reduces the active volume in the reactor, thereby affecting the digester's performance through reduced VS (volatile solids) destruction, which in turn lowers biogas production (Kougias et al. 2013; Rodríguez-Roda et al. 2013; Moos 2012; Shimp et al. 2010).

The following can cause foaming:

- Filamentous microorganisms in the sludge that proliferate during wastewater treatment, most notably Microthrix parcivella, at low-loaded CAS plants.
- Organic overloading of the digester, particularly in cases of high protein and/or ammonia concentrations in the digester feed. This rarely happens in conventional digesters that exclusively digest sludge, but it should be controlled at co-digestion facilities.
- Small sludge surface area in the digester, leading to intensified area-specific gas release.
- The presence of surface active agents in the raw sludge, as well as surface active products of digestion, both of which tend to make the foam more stable.

Operators should be aware that large biogas volumes, as compared to reactor volume, are a major driving force behind foaming. Shimp and others (2010) summarized the situation (figure CS1-3) and compared the phenomenon of digester foaming with "what happens when a carbonated drink is poured into a glass."

Figure CS1-3: Digester gas production rate relative to liquid volume



Source: Shimp et al (2010)

Typical **countermeasures** include the following:

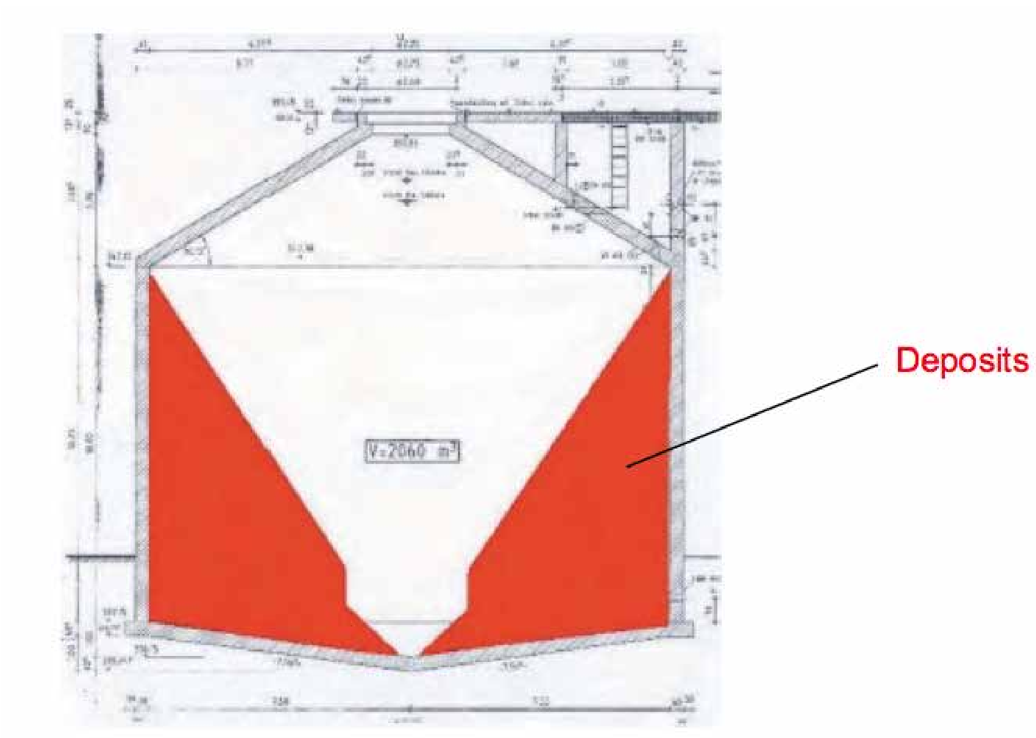
- Preventive, regular (microscopic) control of microbiological characteristics of digester sludge
- Temporary lowering of the liquid level in the digester
- Reduction of organic loading of the digester
- More consistent raw sludge feeding to digesters instead of intermittent feeding in batches
- Utilization of chemical precipitant containing aluminum, by dosing it into the wastewater train for P precipitation and/or by direct dosage into the digester. Poly-aluminum-chlorides (PAC) have become the product most commonly used for this purpose. PAC dosage has proved particularly effective if filamentous microorganisms are causing the foaming. A typical dosage rate is about 1 g Al/kgDS/d.
- Waste activated sludge (WAS) chlorination
- Dosage of chemical defoamers into the digester
- Optimized mixing of the digester
- Installation of foam level detection in the digester, possibly even combining it with automated response (for instance, automatic lowering of the sludge level)
- Installation of special foam destruction installations, such as water nozzles or mechanical devices at the digester's liquid surface

- Installation of sludge disintegration prior to sludge feeding. This proves particularly helpful where the foaming is caused by filamentous bacteria.
- Avoidance as much as possible of physical scum removal from digesters. After all, this measure carries the risk of substantial quantities of biogas being released as well, which in turn increases the risk of explosion (see safety concerns, above).

Deposits in the digester

Deposits in a digester reduce its active reactor volume. Hence, sludge retention time shortens, and VS destruction and biogas production are reduced. Since the volume of deposits can be quite substantial—50 percent reductions are not unheard of—the negative effects on biogas production can also be substantial (see figure CS1-4). In addition, mixing systems can be damaged, and pipes can be clogged. Moreover, the deposits tend to solidify to the point where there is no easy way of removing them. Significant down times of several weeks for digester cleaning and repair may be incurred as well.

Figure CS1-4: Typical shape of deposits in a digester



Source: Heumer 2012.

The following are causes of deposits in digesters:

- Low efficiency of WWTP pretreatment screens. Even standard fine screens with 6 mm free open spacing allow the passage of considerable quantities of solid materials.
- Low efficiency of grit chambers. Any sand passing the grit chambers will inevitably end up in the sludge, and a majority will deposit in the digesters.
- In the case of co-digestion, feeding of solid residues into the digesters that were not efficiently removed from organic waste
- MAP (magnesium ammonia phosphate, also called struvite) formation in the digesters (see figure CS1-5)
- Inefficient mixing systems
- Lack of digester cleaning/maintenance

Figure CS1-5: Struvite deposits from a digester

Source: Ebner 2013.

Typical **countermeasures** include these:

- Installation and proper maintenance of ever finer screens. An open spacing of 6 mm should be seen as the upper limit for any WWTP with digesters. Finer screens of 5 to 3 mm are preferable nowadays.
- Installation and proper maintenance of efficient stages for grit removal
- Installation and proper maintenance of sludge grinders or microstrainers, particularly for primary sludge, prior to sludge feeding into the digesters
- An experts' discussion is currently ongoing on the overall advantages and disadvantages of giving up on enhanced biological P removal technologies from wastewater (bio-P) to minimize MAP (struvite) formation. Dosing Fe into the digester for precipitation of orthophosphate has proved successful in further reducing struvite formation. This also has the positive side effect of H_2S removal from biogas.
- Some utilities have even installed struvite recovery processes to turn the problem into a value-added resource.

Once the problem of deposits manifests (and also in the event that an operator wants to assess a digester's actual condition), there are several ways of approaching the issue:

- Temperature effects. Digesters frequently have temperature sensors both near the bottom and in the upper half. Since deposits usually build up from the bottom, the temperature metered by the bottom sensor differs from that metered by the upper sensor. Ebner (2013) documented a case where the difference between the two sensors amounted to more than 10°C. Such a temperature difference can be taken as an indication of deposits, but it does not allow any quantification of the problem.
- Tracer testing. This is the typical means of quantifying the active reactor volume. A tracer (for example, lithium chloride) is fed to the digester, and its response is analyzed. Two different approaches are possible:

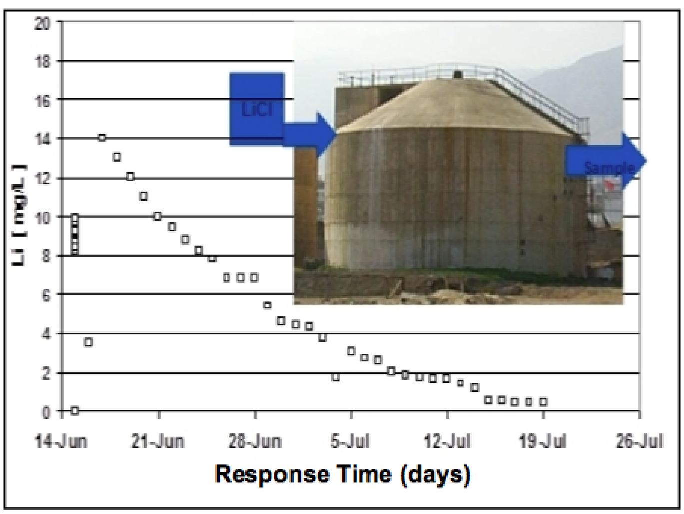
A. The tracer concentration is analyzed in the digester effluent over time. From the characteristics and duration of the tracer response in the effluent it is possible to derive both (a) short-circuiting currents and (b) the active reactor volume of the digester. This type of tracer testing usually takes days or weeks.

B. An exactly defined quantity of tracer is added, the digester is mixed without a feeding of fresh sludge, and the tracer is analyzed in the external circulation pipe that is usually used for mixing fresh feeding sludge with sludge from the digester. From the characteristics and duration of the tracer response it is possible to derive both (a) the time required for complete digester mixing and (b) the active reactor volume of the digester. This type of tracer testing is generally completed within less than twenty-four hours. It is marketed under the name "VoluSense" (Ebner 2013).

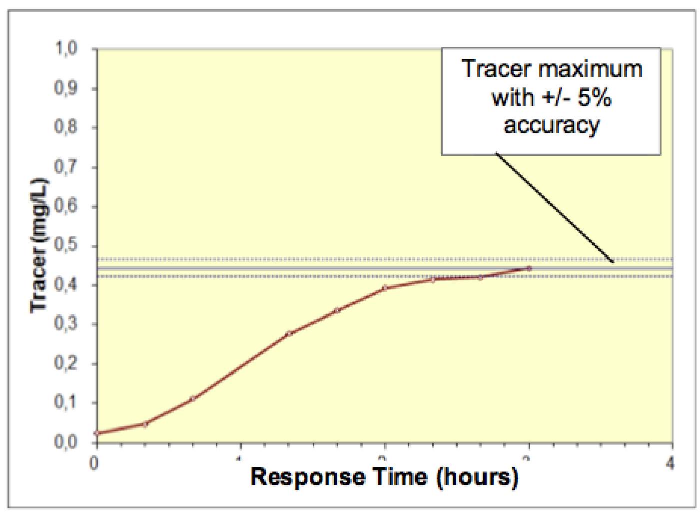
A tracer study of an anaerobic digester with method B costs on the order of US$5,000. Method A, since it takes more time, usually costs between US$5,000 and US$15,000, depending on the size of the digester and the extent of the analysis.

Typical tracer response curves of both methods are presented in figure CS1-6.

Figure CS1-6: Examples of tracer response methods A and B



(Method A)



(Method B)

Sources: Nolasco et al. 2000; ARAconsult 2012.

• Divers. Divers can also be employed to assess the size of deposits in digesters. Several companies have specialized in this field. When selecting a diving company, it is of particular importance to check their specific references. Diving in a digester's environment requires special training, experience, and equipment (see figure CS1-7). Aside from safety considerations, there have been reports of inexperienced divers greatly underestimating the deposits' volume and removal costs (KA-Betriebs-Info Editor 2011). Divers who are professionals in the field, however, have the advantage not only of being able to provide a reliable quantification of the required cleaning work and cost; they can also remove deposits, or at least take samples of them to assess their composition and the difficulties of removal. The disadvantage of divers is that using them typically costs more than a simple reactor volume and mixing analysis. A diver inspection usually costs around US$10,000. Divers specializing in this type of work may not be easily found in EAP.

Figure CS1-7: Divers at work in sludge digesters

Source: www.umwelttauchservice.at.

Once the degree of reduction of active reactor volume and short circuiting are assessed, engineers must decide whether to remove deposits and improve mixing conditions in the digester. The decision of cleaning the digester will depend on multiple factors, including degree of reduction of reactor volume, plant flexibility, and energy generation aspects, among several others. If cleaning is recommended, there are two options for removing deposits:

• Divers. Divers can remove deposits without the whole reactor being emptied. This is sometimes a simple practical necessity, since emptying a digester means managing large sludge volumes in short time periods. It also allows faster removal of the deposits. A typical diver emptying can take about two to three weeks, while complete emptying and restarting can take up to several months (Heumer 2012). However, the cost of diver cleaning is relatively high; it can quickly reach around US$100,000 per digester (Jilg 2012; Heumer 2012).

• Complete emptying of the digester. As mentioned, removing deposits by emptying the digester usually takes more time (generally measured in weeks) than using divers, both for the reactor cleaning itself and for restarting the emptied digester and reaching maximum biogas production again. Emptying also requires a series of auxiliary installations, such as special pumps, suction trucks, cranes, and mobile mechanical sludge dewatering to cope with the sludge qualities and quantities in question. One advantage of emptying is the opportunity for a visual inspection—by the operators themselves—of the pipes and installations located inside the digester, as well as the condition of the concrete walls. If the mixing system is damaged or needs maintenance, the digester can be out of service for a long time. Since emptying and cleaning a digester relies heavily on manpower, its cost will depend on labor costs and digester size. For a fair cost comparison with the use of divers, the cost of extra manpower provided by the operator and the loss of biogas and energy generation during this period must be included in addition to contractor costs.

Insufficient biogas treatment

For the utilization of biogas from digesters, it is necessary to treat the gas prior to its use (for details, see "biogas treatment" in table CS1-5). Foam trapping and condensate removal are standard treatments that must be available at any plant. However, further treatment is sometimes required. When biogas treatment is neglected, it may quickly lead to corrosion problems and to failure of the CHP (see figure CS1-8). Since the necessary repair costs could be high, all too often biogas utilization is then stopped altogether. There are relatively cheap options for biogas treatment that would safely avoid these problems.

As described in Table CS1-5, two main parameters require special attention: H_2S and siloxanes. The table also describes typical threshold values and cost-effective treatment systems and their design. When these technologies are properly applied, the lifespan of biogas utilization can be expected to be long.

To some extent, slightly elevated concentrations of certain substances can be acceptable without treatment. In such cases, shorter maintenance intervals and more frequent lubrication oil changes usually compensate for missing treatments. Such substitutions should always be made in accordance with the CHP suppliers' conditions; otherwise, the operator runs a high risk of being left without a supplier guarantee if damage should occur.

Figure CS1-8: Typical damage caused by insufficient H_2S and siloxane removal: sulfate scaling in a heat exchanger and scaling of siloxanes

Source: VSA 2012b.

1.2.4. Institutional aspects, energy costs

The utilization of biogas in the region of case study 1 (Austria and Germany) is most attractive if the biogas is converted into electric power, since this is a valuable form of energy. It is therefore vital to understand the institutional and legal conditions under which the current boom in biogas production and CHP is taking place. Overwhelmingly, WWTPs in Europe consume the generated electricity onsite. In those rare cases in which a surplus is generated, it is usually sold to the public grid at guaranteed feed-in tariffs that are paid per kWh supplied. A helpful survey of the actual regulations on renewable energy generation in Europe can be found at http://www.res-legal.eu, which also contains all relevant country-specific details.

In Germany, the Erneuerbare-Energien-Gesetz (EEG), the Renewable Energy Sources Act, regulates the supply of electric power to the public grid from renewable sources. It was first introduced in 2000 and has been revised several times since, most recently in 2012. The EEG defines minimum prices per kWh that must be paid (for a period of usually twenty years) to the supplier of electric energy from renewable sources, such as wind-, solar-, hydro-, geothermal-, biomass-, landfill-, and wastewater-based power generation. This law also covers biogas from WWTPs. Grid operators are obliged to give priority to renewable sources when purchasing and transmitting electricity. Additionally, the German Bank KfW is providing low-interest loans, usually with fixed interest rates of about 1 percent per year for ten-year loans, for investments into renewables.

The initial minimum guaranteed feed-in tariff is gradually reduced over the years. The intention is to promote renewables strongly in the beginning, but also to make them more efficient over time. Electricity supplies from WWTPs to the grid receive EUR0.0589–0.0679/kWh (US$0.080–0.092/kWh). This is generally less than 50 percent of the unit electricity cost the WWTPs have to pay when purchasing from the public grid. Consequently, onsite generation and utilization of electricity from biogas at the WWTPs is more attractive than supplying it into the public grid.

Austria also has a legally defined feed-in tariff for electricity from renewable sources. Details are defined in Ökostrom-Einspeisetarifverordnung 2012 (ÖSET-VO 2012, By-Law on Renewable Electricity Feed-in Tariff). In 2014, the feed-in tariff for electricity produced from WWTP biogas equaled EUR0.0594/kWh (US$0.080/kWh), fixed for a period of thirteen years.

Other issues and conclusions related to German and Austrian regulation in 2014 include the following:

1. The overwhelming majority of WWTP operators utilize power from biogas to cover their own electricity needs onsite. Supply to the public grid is considered only in rare cases of surplus energy. Usually the latter is not financially attractive due to low feed-in tariffs.

2. The feed-in tariffs for electricity from other resources (for instance, solar and wind) are considerably higher than those for biogas-based electricity and can go up to about EUR0.20/kWh (US$0.27/kWh) in both countries.

3. Domestic consumers currently pay an average electricity tariff of about EUR0.20/kWh (US$0.27/kWh) in Austria and EUR0.26/kWh (US$0.35/kWh) in Germany.

4. Industrial consumers, including WWTPs, pay lower electricity tariffs than domestic consumers. WWTPs are typically charged about EUR0.10/

kWh (US$0.135/kWh) in Austria and EUR0.15/kWh (US$0.20/kWh) in Germany.

5. When the period of guaranteed feed-in tariffs ends, WWTPs usually have to cope with substantial reductions of their feed-in tariffs. Feed-in tariffs could be as low as EUR0.02–0.03/kWh (US$0.03–0.04/kWh). Only then do alternatives to grid supply for (rare) surplus energy, such as supply into natural gas pipelines, become attractive.

1.2.5. GHG reduction and CDM co-financing

General

European climate policy targets a drastic reduction of greenhouse gas (GHG) emissions. To boost renewable energy and minimize energy inefficiencies, the European Union (EU) uses three instruments:

- At the EU level, it defines emissions targets. Just recently, on January 22, 2014, the new targets were announced by the EU Commission: (a) 27 percent of total energy is supposed to come from renewables by 2030 for the EU as a whole; (b) 40 percent reductions of GHG emissions until 2030, as compared to emissions in 1990.
- At the national level, it provides financial incentives and subsidies to achieve the emissions targets (for instance, as described for WWTPs in the section on institutional aspects, above).
- It uses a trading scheme for GHG emissions to introduce a fair price for pollution.

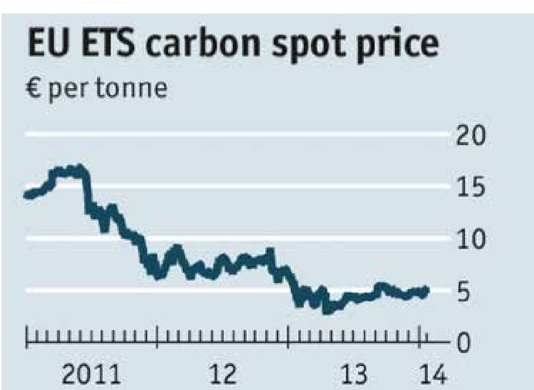
The EU's emissions trading scheme (ETS) operates under the Kyoto Protocol via the Clean Development Mechanism (CDM), by which countries that have ratified the Kyoto Protocol can invest in projects that reduce GHG emissions in developing countries. These investments can be traded to signatory countries, which can use these certified emissions reductions (commonly known as carbon credits) to meet their commitments under the Kyoto Protocol.

At present, 18 percent of projects registered as CDM worldwide are based on anaerobically digested biogas, with Brazil, Malaysia, Mexico, and the Philippines the countries with the most projects (Chamy 2013). The majority of projects originate from the industrial sector and not from digesters at municipal WWTPs.

Price of carbon credits

The basic idea of the emissions trading scheme is that GHGs are reduced where the cost to do so is lowest. However, the EU market was flooded with an excessive number of permits, so the price decreased, and in 2014 it fell to a low of about EUR5/ton CO_{2e} (US$6.8/ton CO_{2e}); see figure CS1-9.

Figure CS1-9: EU ETS carbon spot price



NOTE:

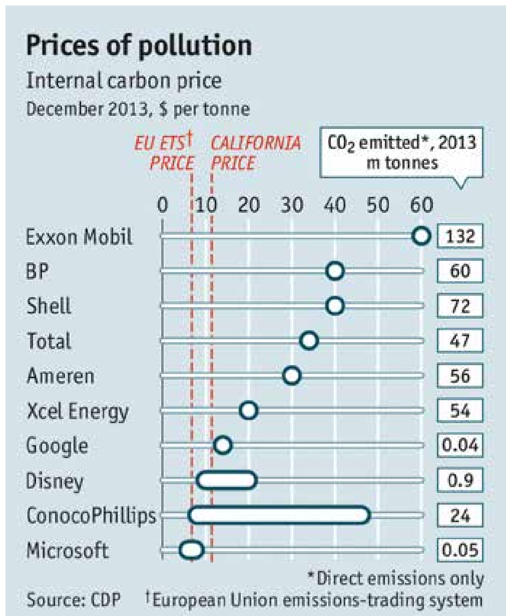
The EU ETS is just one specific emissions trading scheme prevailing in the region of case study 1. Besides ETS, there are others, and prices may vary from scheme to scheme. Notwithstanding, what all schemes have in common at present are (very) low prices per ton CO_{2e}, some being even lower than those of ETS.

Source: The Economist 2014.

As a consequence, the actual cost for cutting emissions is considerably higher due to the combined effects of subsidies. *The Economist* (2014) sets the cost at over EUR150/ton CO_{2e} (US$200/ton CO_{2e}) under the renewables program. Therefore, co-financing through ETS is not attractive for the time being.

A change would be possible if the real cost were, indeed, reflected by the cost of carbon credits. It would be an indication of what the real cost is when assessing companies' internal risk calculations. Many companies use an "internal carbon price" per ton of CO_{2e} for planning purposes (see figure CS1-10).

Figure CS1-10: Internal carbon price of selected companies



Source: CDP = Carbon Disclosure Project (UK-based NGO), cited in The Economist *2013.*

The internal prices range from US$6/ton CO_{2e} at Microsoft to US$60/ton CO_{2e} at Exxon Mobil. A particularly close look at those companies that emit large amounts of GHG seems to indicate that a price of about US$40/ton CO_{2e} reflects reality better than the results from ETS. If this price were to materialize, CDM co-financing could generally, and also specifically for WWTPs, become a much more appealing instrument in the future.

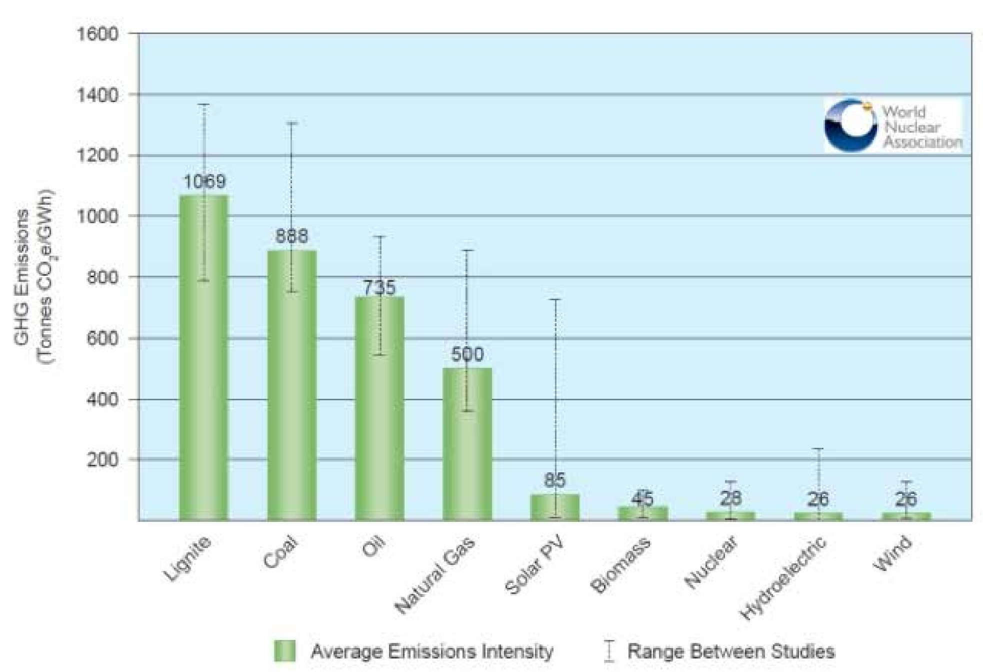
Relevant aspects for CDM projects

Under the principle of additionality, GHG emissions from a CDM project must be reduced below those levels that would have occurred in the absence of the project. This poses uncertainties to any CDM project at WWTPs involving biogas utilization from new digesters. After all, without the digesters, and therefore without the project, there would be no methane that could be reduced. Since methane is a stronger GHG than CO_2, already relatively low levels of methane utilization bring substantial carbon credits.

A second issue is the question of how to calculate the potential reduction in CO_{2e} for a digestion project at a WWTP. Two main approaches can be applied individually or combined: reduction of electricity generation from fossil fuels and reduction of methane emissions.

Reduction of electricity generation from fossil fuels is the classical approach when a WWTP is equipped with a new digester and a new CHP for the utilization of biogas. Since the sludge is considered a renewable, any electricity generation from a sludge digester's biogas is from renewable sources. This consideration is not limited to the electricity generation from biogas alone. It also includes indirect impacts of the digester, such as reduced electricity consumption in (already existing) aeration tanks, where extended aeration is no longer required, since the sludge will be stabilized in the digesters after project implementation. The total of electricity savings plus electricity generation implies a reduced need for that specific amount of electricity generated from fossil fuels. The question is, then, how much of GHG emissions are caused by electricity generation from fossil fuels? This depends on a country's specific energy matrix. Different fossil fuels imply different GHG emissions in electricity production (see figure CS1-11). Technology specifics and country specifics influence the respective GHG emissions for electricity production.

Figure CS1-11: Specific GHG emissions of electricity generation from fossil fuels in the world



Source: World Nuclear Association 2011.
Note: 1 ton CO_{2e}/GWh = 1 g CO_{2e}/kWh

Table CS1-3: GHG emissions per kWh for electricity generation in Germany and Austria, compared to European Union and the world

REGION	1990 gCO_2/kWh	2010 gCO_2/kWh
Germany	607	**461**
Austria	238	**188**
EU (27 member countries)	585	**429**
World	586	**565**

Note: The table shows CO_2 emissions from fossil fuels consumed for electricity generation, in both electricity-only and combined heat and power plants, divided by output of electricity generated from fossil fuels, nuclear, hydro (excluding pumped storage), geothermal, solar, wind, tide, wave, ocean, and biofuels. Both main activity producers and autoproducers have been included in the calculation.

Source: IEA 2012.

As it turns out (see table CS1-3), GHG emissions are rather different, depending on the composition of a country's or region's energy production matrix. These specifics always have to be considered in a CDM project.

Reduction of methane emissions can usually be applied with existing WWTPs, where methane is already being emitted into the open air and the CDM project is introducing new installations that mitigate those emissions. Simple collection of the methane and subsequent flaring can render a project eligible for the CDM mechanism. However, the approach may also include a component of electricity generation from the collected methane, which further increases its potential for carbon credits. For instance, the covering of an open sludge digester (but also of any other stages, such as anaerobic ponds) and capture of the emitting methane would fall into this category.

As pointed out before, due to the additionality criterion, the new construction of a digester complete with CHP only takes into account the energy production component under CDM, but not the methane elimination, since the methane is created and eliminated by the project itself. An exception to this would apply in countries where business-as-usual conditions are to release biogas to the atmosphere without burning it. For example, in Bolivia, most anaerobic ponds are not covered, hence the baseline in this country is to release methane generated in such ponds to the atmosphere. In Bolivia, covering the ponds to capture biogas would be additional to business-as-usual (baseline situation). Thereby, mitigation of methane by collection and burning can be considered in the CO_{2e} calculation, in addition to any other credits originating from electricity generation with biogas, which displaces generation emissions elsewhere.

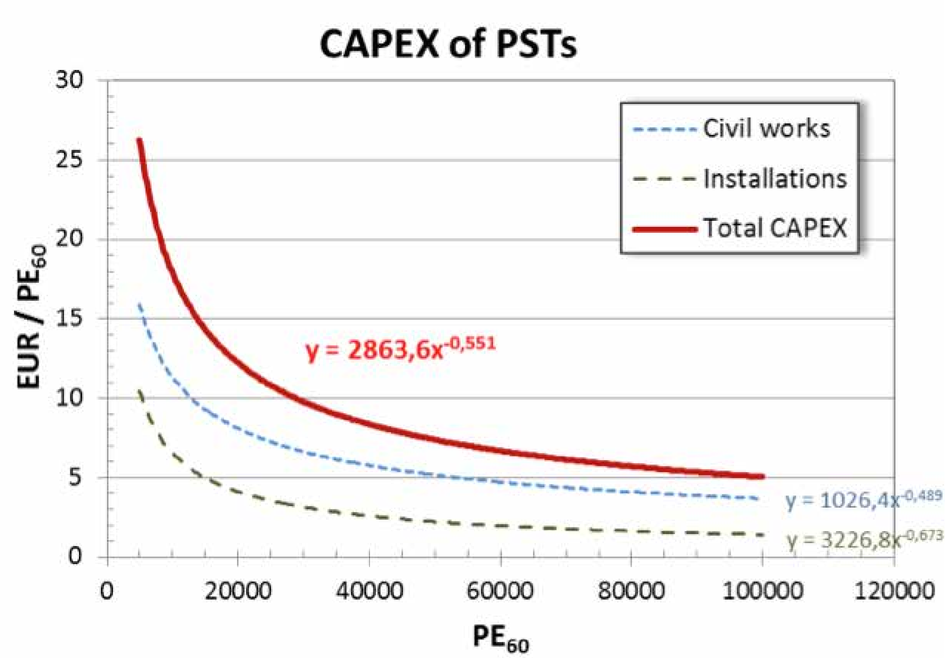
1.2.6. CAPEX structure

Up-to-date cost curves have been developed for CAS (Tectraa et al. 2010, 2011; Gretzschel et al. 2012) reflecting the actual costs of all prevailing elements for sludge digestion in the area of case study 1 when switching from extended aeration to mesophilic sludge digestion. The elements of the cost analysis include the following:

- CAPEX of primary sedimentation tank (figure CS1-12)
- CAPEX of anaerobic digester (figure CS1-13)
- CAPEX of all other concerned items (intermediate pumping station, primary sludge pumping station, primary sludge buffer tank, mechanical sludge thickener, gas holder, gas flare, CHP, operation building, pipes, traffic areas, other costs; see figure CS1-14)

Figure CS1-15 presents the overall results from all three items together.

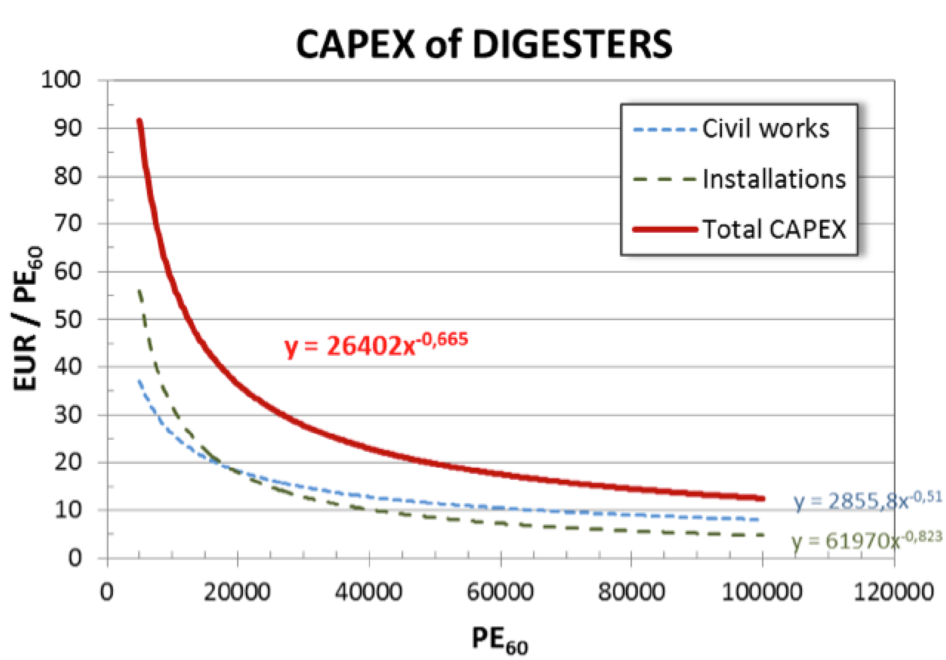
Figure CS1-12: Specific CAPEX for primary sedimentation tanks



Source: Tectraa et al. 2010, 2011; Gretzschel et al. 2012.

Notes: CAPEX includes engineering and 19 percent value-added tax (VAT). EUR/PE$_{60}$ x 16.67 = EUR/kg BOD$_5$.

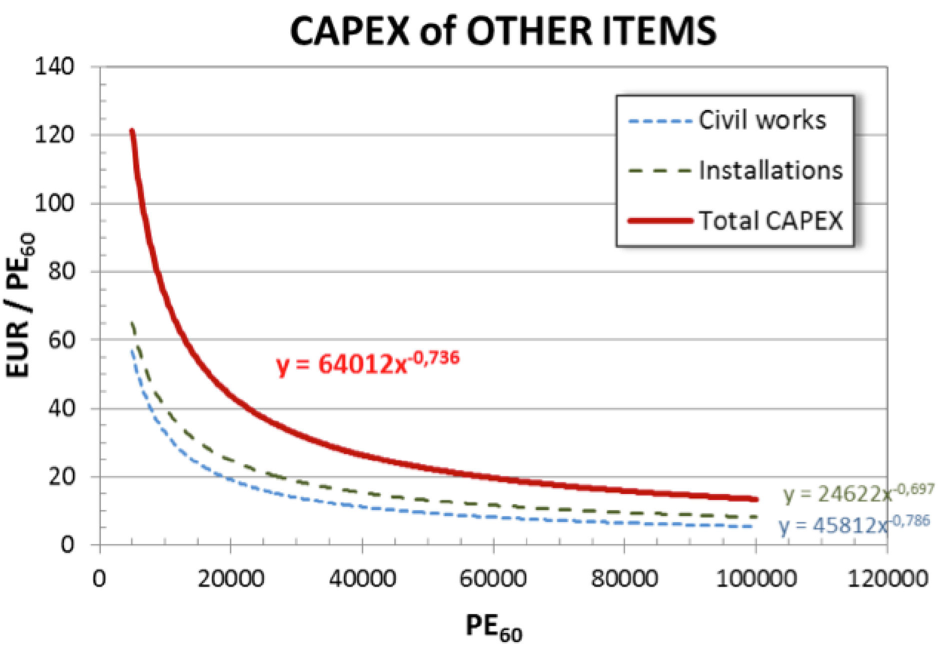
Figure CS1-13: Specific CAPEX for Sludge Digesters



Source: Tectraa et al. 2010, 2011; Gretzschel et al. 2012.

Notes: CAPEX includes engineering and 19 percent value-added tax (VAT). EUR/PE$_{60}$ x 16.67 = EUR/kg BOD$_5$.

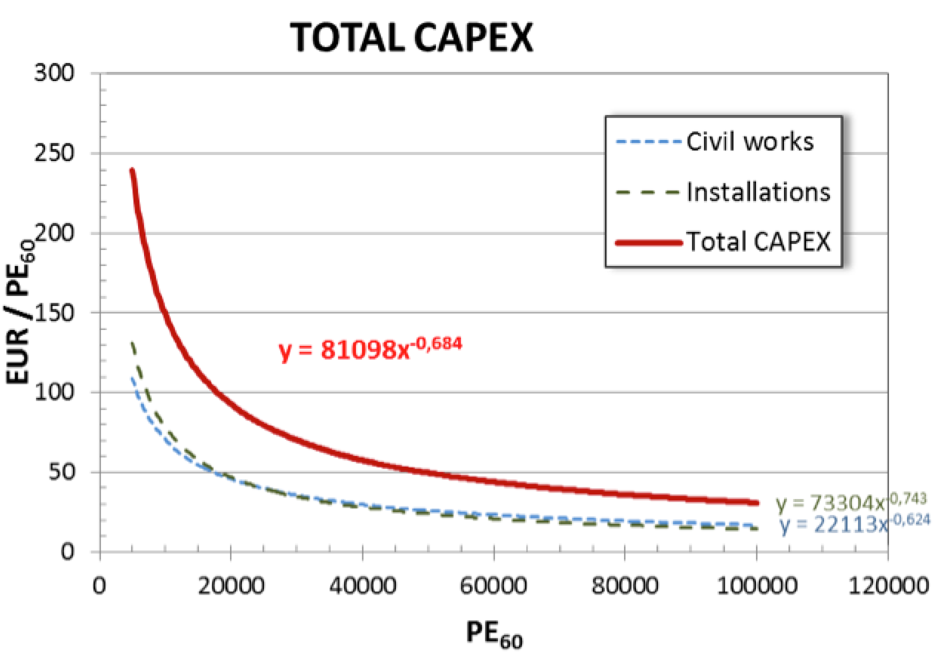
Figure CS1-14: Specific CAPEX for all other items concerned



Source: Tectraa et al. 2010, 2011; Gretzschel et al. 2012.

Notes: CAPEX includes engineering and 19 percent value-added tax (VAT). EUR/PE_{60} x 16.67 = EUR/kg BOD_5.

Figure CS1-15: Specific total CAPEX for technology change to mesophilic sludge digestion

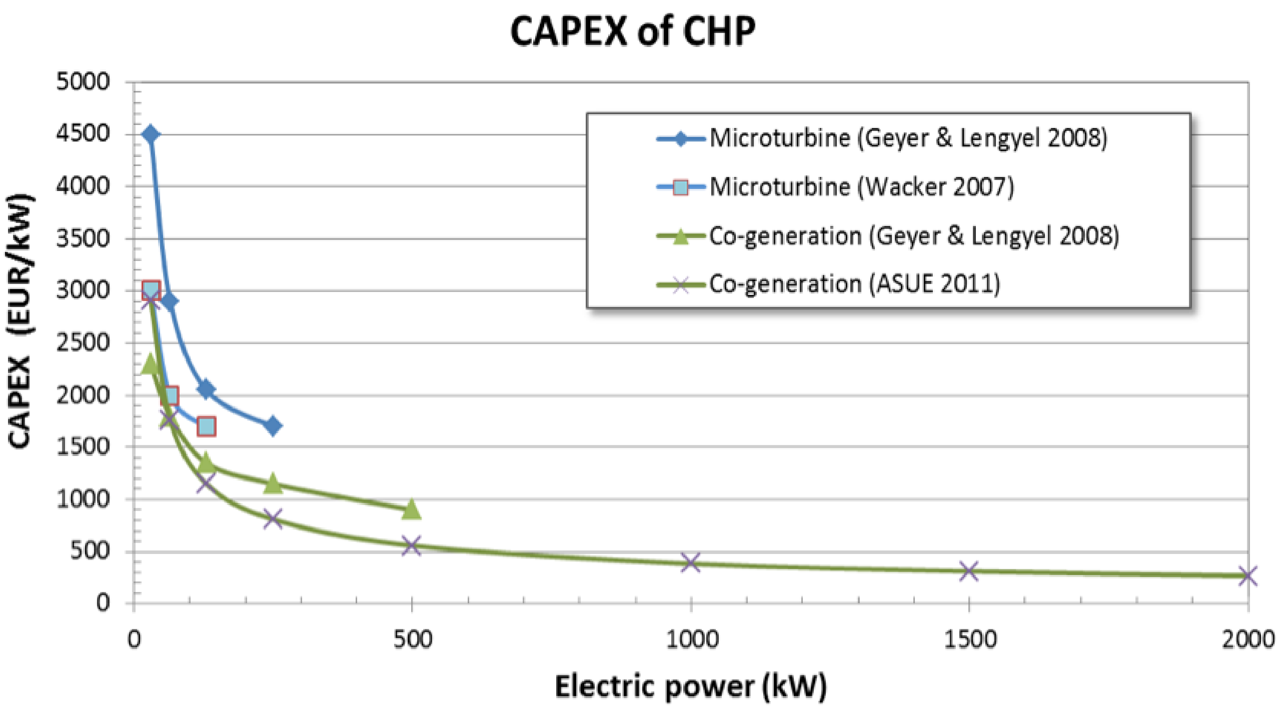


Source: Tectraa et al. 2010, 2011; Gretzschel et al. 2012.

Notes: CAPEX includes engineering and 19 percent value-added tax (VAT). EUR/PE_{60} x 16.67 = EUR/kg BOD_5.

Since the preceding figures do not explicitly present CAPEX for CHP, figure CS1-16 summarizes CHP cost, as indicated in different sources.

Figure CS1-16: CAPEX for microturbines and co-generation



Source: Wacker 2007; Geyer and Lengyel 2008; ASUE 2011.

WERF (2010b) mentions an estimate for CAPEX of US$4,124/kW (EUR3,050/kW) for microturbines. No indication of the electric capacity behind this value is provided. Nonetheless, the CAPEX level of microturbines seems similar between the United States and Europe.

1.2.7. OPEX structure

The same cost basis (Tectraa et al. 2010, 2011; Gretzschel et al. 2012) that was presented in the previous section for CAPEX is used for the OPEX assessment of case study 1. The following cost items are distinguished when introducing mesophilic sludge digestion.

OPEX increase:

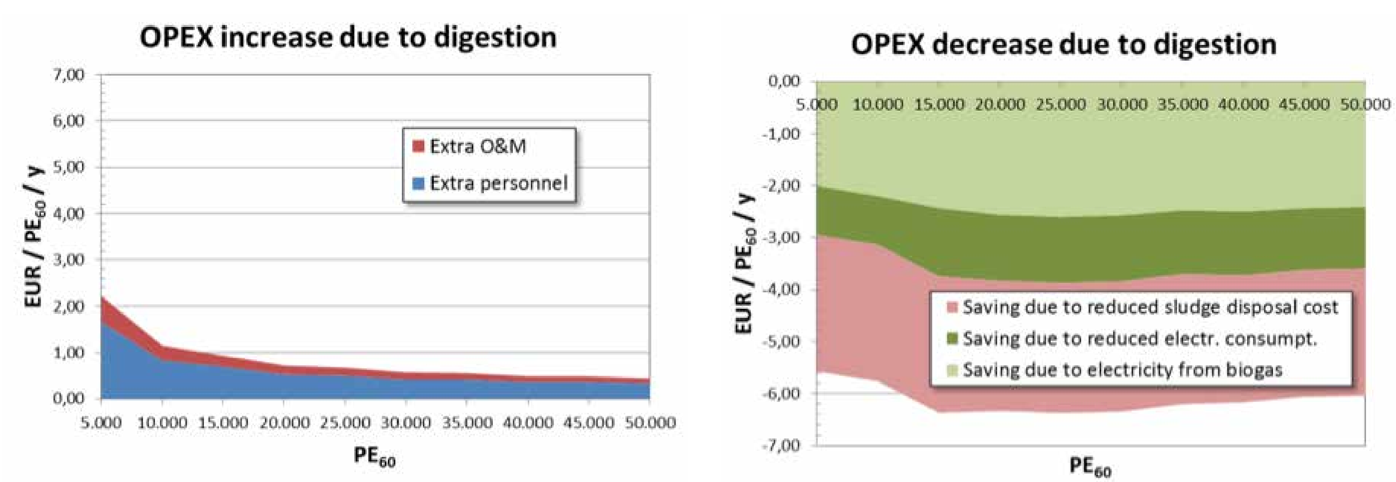
- Extra OPEX of operation and maintenance (O&M) of additional installations
- Extra OPEX of extra personnel needed to operate additional installations

OPEX decrease:

- OPEX savings due to reduced sludge disposal cost, brought about by (a) stronger degradation of volatile solids in digester as compared to extended aeration and (b) better dewatering properties of digested sludge
- OPEX savings due to reduced electricity consumption
- OPEX savings due to use of electricity from biogas to reduce electric power purchases from the public grid

Figure CS1-17 presents the quantified OPEX increases and decreases under the specific conditions in Germany. Figure CS1-18 presents the overall OPEX reduction.

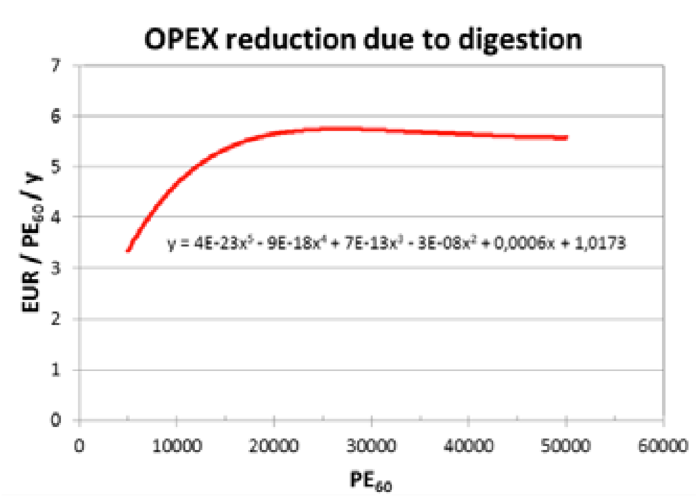
Figure CS1-17: OPEX increase and decrease caused by mesophilic digestion and biogas utilization



Source: Tectraa et al. 2011.

Notes: CAPEX includes engineering and 19 percent value-added tax (VAT). EUR/PE_{60} x 16.67 = EUR/kg BOD_5.

Figure CS1-18: Overall OPEX reduction caused by mesophilic digestion and biogas utilization



Sources: Tectraa et al. 2011; Gretzschel et al. 2012.

Notes: CAPEX includes engineering and 19 percent value-added tax (VAT). EUR/PE_{60} x 16.67 = EUR/kg BOD_5.

An overall conclusion is that OPEX could be reduced by EUR5–6/PE_{60}/y (in Germany or Austria) when introducing mesophilic digestion and biogas utilization, as compared to OPEX of extended aeration.

The typical median total OPEX of CAS WWTPs in the region varies between EUR11 and EUR21/PE_{60}/y for large WWTPs of more than 100,000 PE_{60} and for plants of 10,000–50,000 PE_{60}, respectively (OEWAV 2012). Hence, the introduction of sludge digestion and biogas utilization reduces total OPEX of the plants concerned by about 15–50 percent.

As can be observed from figure CS1-17, O&M cost of the new installations is not significant. Nonetheless, the relative importance of this item could be different in other regions. Supplementary data are presented below concerning the respective O&M costs of biogas treatment and of CHP.

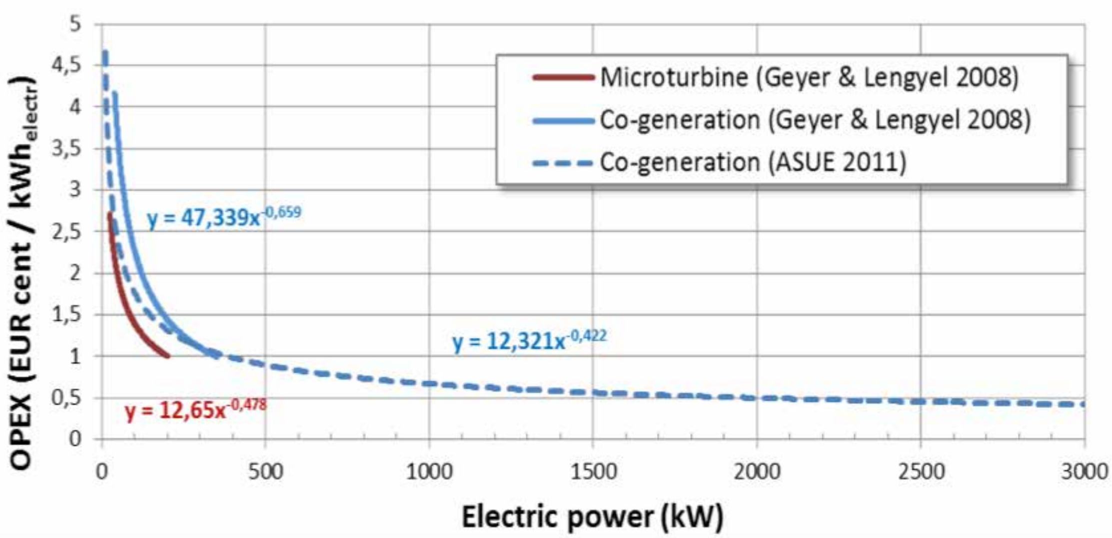
Biogas treatment

OPEX of biogas treatment can vary within a wide range, depending on untreated biogas quality, treatment (CHP) requirements, and treatment technology. A typical cost range is between EUR0.005 and EUR0.050/m^3 biogas (EUR0.02–0.5/PE_{60}/y, based on biogas production rates from 10 to 30 L/PE_{60}/d, according to figure CS1-2). Micro-aeration or Fe dosage are on the lower end of that range, and activated carbon adsorption is usually in the middle; and the upper end of that cost range—sometimes even beyond it—is required for cleaning and supplying biogas into natural biogas supply pipelines.

CHP

Many suppliers of CHP offer service contracts. Their cost offers a perfect indicator of the actual OPEX involved. ASUE (2011) analyzed sixty-one contracts and derived cost curves for co-generation, while Geyer and Lengyel (2008) developed similar curves for co-generation and microturbines. VSA (2012b) and Wacker (2007) cite such costs as well, confirming the more detailed cost information from Geyer and Lengyel and ASUE. The results are summarized in figure CS1-19.

Figure CS1-19: OPEX of CHP



Source: Geyer and Lengyel 2008; ASUE 2011.

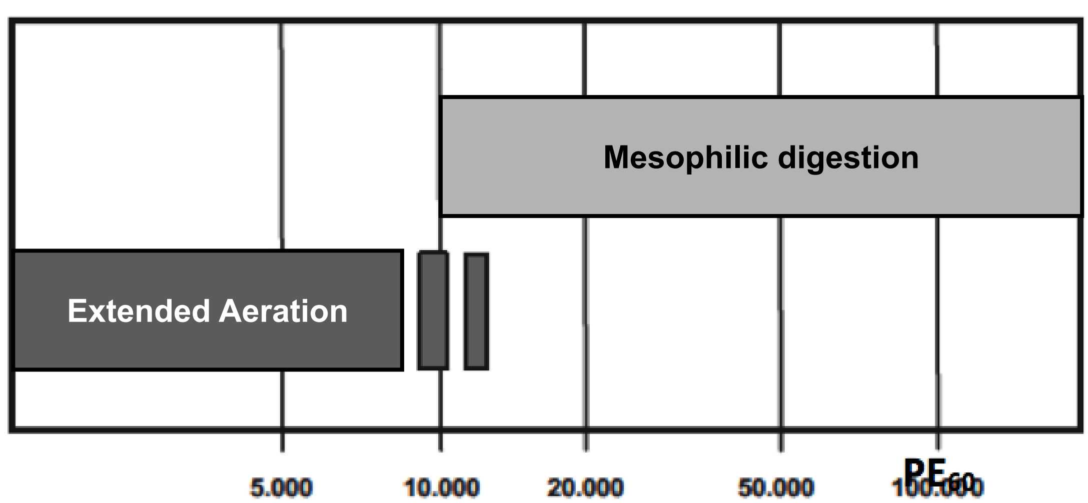
WERF (2010b) mentions OPEX of US$0.02/kWh (EUR0.015/kWh) for microturbines. Although no indication of the electric capacity behind this value is provided, the OPEX levels of microturbines in the United States and Europe seem similar.

1.2.8. Viability of investment in biogas utilization

Project financial viability is usually evaluated by looking at life cycle cost. The lifespans of the various components differ, with civil works having a longer lifespan (typically thirty years) than electromechanical equipment and installations (typically ten to fifteen years). When weighing the relative extent of the various cost elements, an average lifespan of roughly fifteen to twenty years can be assumed. Viability thus means that the financial return on the investment within those fifteen to twenty years is larger than the cost.

The savings in operating costs can be estimated via an average savings of some EUR5–6/PE_{60}/y, as derived in figure CS1-18. Within a period of fifteen to twenty years, this amounts to EUR75–120/PE_{60}. This amount is equivalent to CAPEX required for introducing digestion + biogas utilization for a WWTP size of 15,000–25,000 PE_{60} (see figure CS1-15). Generally, that means that in the region of case study 1, WWTPs with a capacity of more than 15,000–25,000 PE_{60} are financially more viable if they include digesters and biogas utilization. This is also what is currently concluded by various authors (Tectraa et al. 2010, 2011; Gretzschel et al. 2012; Dohmann and Schröder 2011). If the electricity unit cost increases further in the future, this financial threshold is expected to decline to about 10,000 PE_{60} (see figure CS1-20).

Figure CS1-20: Future viability of mesophilic digestion versus extended aeration



Source: Dohmann and Schröder 2011.

1.3. Conclusions for CAS + sludge digestion in EAP countries

In EAP, the characteristics and quantities of sludge that can be expected from a CAS plant may be different than in Europe. This change will become particularly evident where wastewater dilution is high and where septic tanks continue to be used in large numbers. These effects imply a shift toward a higher percentage of WAS, compared to PS. This, combined with reduced total sludge quantities, will inevitably lead to reduced biogas production, and thus to a reduced electricity generation potential.

CAPEX of digesters may be lower in EAP than in Europe; civil works in particular may be cheaper. Given that about 50 percent of total CAPEX is for electro-mechanical equipment, which is mostly imported and hence not much cheaper, the overall CAPEX reduction for a digester system in EAP as compared to Europe could be around 20 percent.

On the other hand, OPEX savings will be lower in EAP in absolute terms. Sludge disposal cost is usually very low as compared to Europe, and in most cases the unit electricity cost is lower, as well. Thus, the financial value of the generated electricity is also lower.

The reduced CAPEX will consequently have to be balanced with reduced OPEX savings. Therefore, the amortization periods of investment into sludge digestion and biogas projects in EAP are similar to or somewhat longer than in Europe.

In terms of electricity coverage, a CAS plant + digester in EAP can usually be expected to be capable of producing only part of the electricity required for the operation of that facility. Plants designed for carbon removal may achieve a higher coverage of up to 50–75 percent, whereas plants designed for nitrification/denitrification are expected to achieve a lower coverage of just 30–50 percent.

Success stories in EAP

Successful examples of CAS + sludge digestion projects are so far mainly limited to large cities and WWTPs. For instance, Cao (2011) has reported on the Ulu Pandan Water Reclamation Plant in Singapore, which is based on CAS and achieving 63 percent N removal.

It is currently operating close to its design capacity of 361 MLD. The influent is normal strength wastewater with TSS (total suspended solids) of about 300 mg/L and COD (chemical oxygen demand) of about 600 mg/L. This plant generates 22,424 m^3 biogas/d, equivalent to about 12.2 L biogas/PE_{60}. This is on the lower end of the biogas expectation range indicated in figure CS1-2, but it still shows the digestion process works properly in EAP. Cao reported that through optimization measures, the plant is expected to increase the present electricity coverage from 34 percent to about 55 percent of its total needs.

An interesting analysis of sludge management practices and options in China was prepared by ADB (2012). It found that more than 80 percent of sludge is disposed of at landfills without prior stabilization. The official policy tries to change this practice through improved volume reduction, stabilization, and safe disposal. Still, for the time being, the number of anaerobic sludge digesters with biogas production/utilization is small. A few examples stand out, such as sludge digesters at Sanjintan WWTP in Wuhan (Hubei Province), or Sibao WWTP in Huangzhou (Zheijang Province).

Another example is the Bailonggang Municipal WWTP in Shanghai, China, with a design capacity of 2,000 MLD and 4.3 million PE, featuring eight large, egg-shaped digesters with a total volume of 99,200 m^3; when it was successfully commissioned in 2011, it was said to be the largest such sludge treatment facility in the world. This plant is receiving thin wastewater with just about 100 mg BOD_5/L and 100 mg TSS/L, the latter being overwhelmingly inorganic (Enviro-Consult and Sogreah China 2007). The biogas production is working well, with a yield of 0.82 $m^3/kgVSS_{added}$. Operational problems have been observed related to foaming and deposits inside the digesters (Jiang et al. 2013). About 50 percent of the actual biogas production of about 33,000 m^3/d is used for sludge drying and digester heating, while the remainder is currently just flared (private communication 2014).

An interesting case study was done for four WWTPs in Chengdu, China, by Murray and others (2008). They analyzed a wide range of sludge treatment options, including dewatering, lime addition, mesophilic anaerobic digestion, heat drying, and incineration, and various combinations thereof. Based on a life cycle assessment, they concluded that “anaerobic digestion is generally the optimal treatment.” Even though the authors based their analysis on the Chengdu case study, they concluded the outcome should be representative for many other WWTPs worldwide.

In Vietnam, Yen So WWTP in Hanoi (200 MLD) features sludge digestion, as well. It apparently worked well during commissioning and was producing biogas. Digester operation was then stopped, however, due to unresolved contractual issues between the private operator and the project owner.

Some of the few existing sludge digesters in East Asia are suffering from problems associated with foaming and deposits in the digester. It is assumed that this is typically the consequence of inadequate preliminary treatment (screening, grit removal). To avoid such problems in the future, it may be worthwhile to consider DBO (design-build-operate) contracts for WWTPs with sludge digesters. Under such a contract, experienced private companies in charge of the complete WWTP design and operation would be expected to install proper components at all stages. As case study 2, below, demonstrates, such an approach can work well.

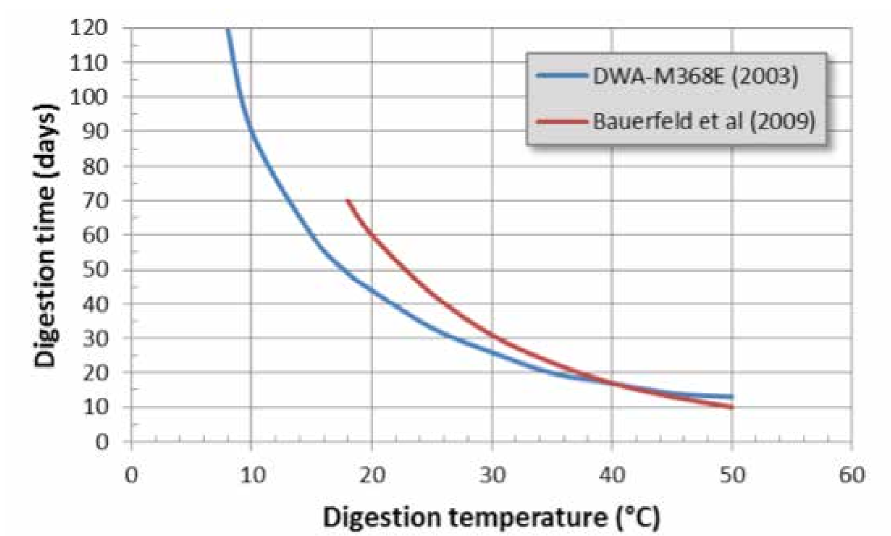
Table CS1-4: Anaerobic digesters: General design parameters and key characteristics

DIGESTER DESIGN

Retention time: Fifteen to twenty-five days for large–small WWTPs at mesophilic temperature 30–38°C (VSA 2012a; WERF 2010c; DWA 1996). Note that some guidebooks recommend shorter retention times down to about ten days (for instance, Metcalf and Eddy 2003), but this only serves to cover the most intensive period of biogas production; it is insufficient to stabilize the sludge properly under all operation conditions.

With unheated digesters, the annual fluctuation of air temperature and its impact on digestion/biogas production should be analyzed carefully. In East Asia and other regions, biogas production can be successful even when temperature minimums go down briefly to about 25°C. Figure CS1-21 presents the digestion time requirements as a function of temperature. This allows for the necessary digestion time in case of non-mesophilic conditions.

Figure CS1-21: Anaerobic digestion time required to achieve "stabilized sludge" in function of temperature



Source: DWA 2003; Bauerfeld et al. 2009.

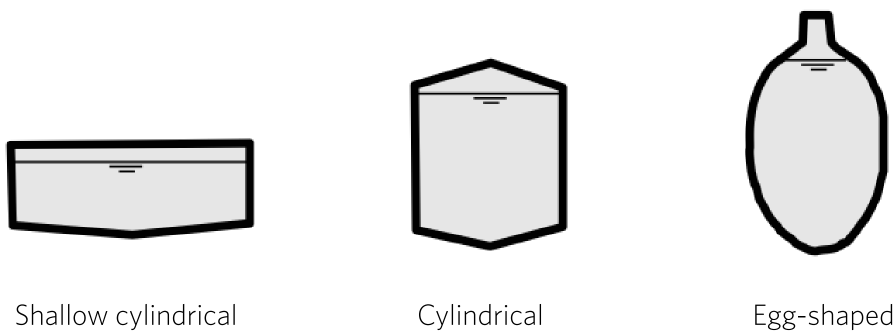
Bauerfeld and others (2009) also highlight that unheated digesters are a research focus in Vietnam at Ho Chi Minh City's universities (Nong Lam University; University of Technology), which are considering this as a meaningful technology, particularly for large WWTPs.

Given the long retention time of sludge in digesters, designers should avoid calculating sludge quantities for one-day peaks and apply those quantities to digester design. Rather, two-week or one-month peak values are adequate for digester design, since the typical twenty-day criterion already includes a safety margin of about five days against brief sludge peaks (for instance, caused by stormwater peaks, feeding only on five days per seven-day week, or unreliability of sludge production forecasts). Most of the biological degradation takes place within the first ten days (see figure CS1-25).

Solids loading rate: ($kgVS/m^3/d$) is sometimes recommended as a design parameter. However, it never prevails for digesters fed with primary sludge and WAS. Only in cases of co-digestion of additional organic waste does this parameter deserve more attention (see case study 5). Typical maximum permitted loadings are 1.5–4.5 $kgVS/m^3/d$ for small–large WWTPs at mesophilic temperature 30–38°C (VSA 2012a; Metcalf and Eddy 2003; Schmelz 2000; DWA 1996).

DIGESTER SHAPE AND CONSTRUCTION

Figure CS1-22: Typical shapes of anaerobic



Source: Authors.

Temperature control and adequate mixing are key to achieving adequate performance in anaerobic digesters. Usually, shallow digesters result in poor mixing, which in turn fosters grit settlement and accumulation of scum. On the other hand, the large surface area of cylindrical shapes provides for less intensive biogas emissions per unit area, which reduces the risk of scum formation. As inert material accumulates in the digester, the active volume is reduced, thereby reducing its performance. Reduced anaerobic digestion performance affects biogas production and subsequent energy generation.

Egg-shaped digesters have the lowest outer surface to volume ratio and thus the minimum energy losses, which matters particularly in cold climate regions. They also feature good mixing properties, but they are more difficult to construct and usually require higher CAPEX than cylindrical shapes.

Construction is mostly done in reinforced concrete, but steel construction is sometimes used. The latter usually requires somewhat less CAPEX but is considered riskier in terms of corrosion and bursting.

The state-of-the-art types of digester covers include floating, fixed, and membrane. The most common application is a fixed cover that provides free space between the digester roof and the liquid surface. The membrane cover is a relatively new development that combines its cover function with a gas holder on top of the digester. This may be attractive where land is in short supply and can offer cost advantages.

Figure CS1-23 presents a combination of cylindrical and egg-shaped digester.

Figure CS1-23: Cylindrical and egg-shaped digester

Source: Fritzens WWTP, Austria.

DIGESTER MIXING

Mixing serves various purposes: (a) bringing substrate in contact with active biomass; (b) avoidance of sludge stratification and scum layer; (c) improved gas stripping; and (d) avoidance of dead volume. The following are the most widespread systems for digester mixing:

- Mechanical low-speed mixers inside the digester
- Biogas injection
- Mechanical pumping with internal draft tube
- Mechanical pumping with external pumps

Different mixing systems have different efficiencies. No systematic comparison exists, however, and specific analysis is only sometimes done. For instance, Jenicek (2012) reports that replacing biogas injection with mechanical low-speed mixers inside Prague's digesters increased actual sludge retention time by 25 percent. Careful analysis is hence always recommended before deciding on a specific mixing system.

DWA (1996) recommends that mixing systems have a daily mixing capacity of about 12–15 times the digester volume. Higher values are not deemed necessary as, for instance, recommended by Metcalf and Eddy (2003), who define a turnover time of tank contents of 20–30 minutes, which would imply a daily mixing capacity of 48–72 times the digester volume.

In general, mixing is not required twenty-four hours a day, but it should always be done during feeding of raw sludge.

DIGESTER FEEDING

To promote fast and intensive contact of fresh sludge with active biomass in the digester, it is common practice to mix digester sludge with raw sludge and to feed the mixture into the digester. Some authors recommend mixing ratios of 2–4:1 between digester sludge and raw sludge, with a ratio of 1:1 considered the minimum requirement.

Generally, to minimize sludge sedimentation in pipes it is recommendable to consider minimum values, for instance, DN U 100 mm and v U 0.8 m/s (DWA 1996).

The sludge mixture usually passes through a heat exchanger before being injected into the digester.

DIGESTER HEATING

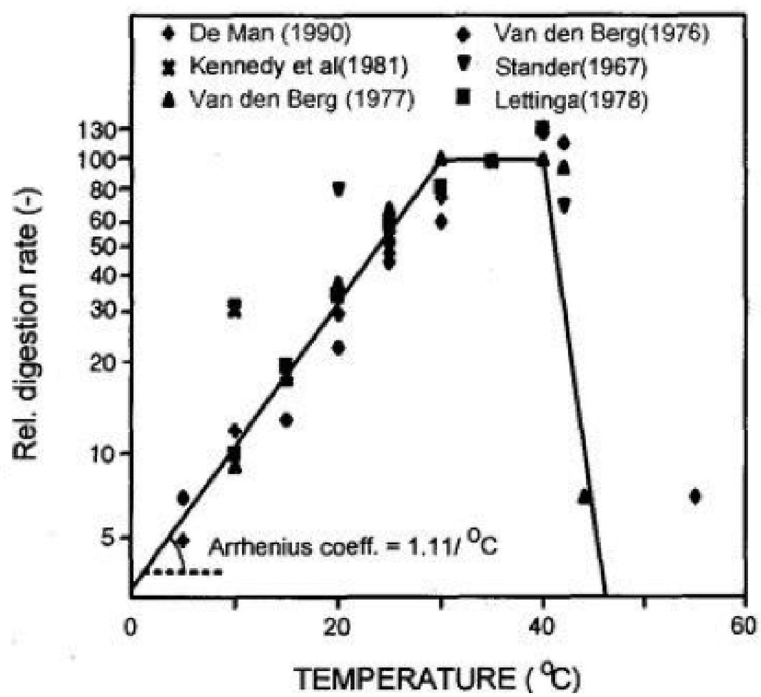
Digester heating serves two purposes:

- Heating of (cold) raw sludge to mesophilic temperature
- Compensation of heat losses through the walls, floor, and roof of the digester

It is most common to employ external heat exchangers. In some cases internal heating devices inside the digester are also in use, but the maintenance of these units can prove problematic; therefore, internal heating is not a standard solution. The two most common types of external heat exchangers are heated tube-in-tube and spiral plate heat exchangers.

In developed countries with cold winters, the thermal energy produced from CHP (about 50 percent of the biogas's calorific value) is mostly sufficient to enable digester heating without additional external fuels. Only very cold periods may require additional heat sources. Consequently, in warm climates heating never becomes a real OPEX issue. It has even proved possible in warm climate countries to eliminate digester heating and digester insulation (see case study 2). In this case, it is important to ensure a minimum temperature in the digester, ideally >28–30°C. Below that value the biological activity reduces drastically (see figure CS1-24). Then, if lower temperatures persist for prolonged periods, neither biogas production nor sludge stabilization will be satisfactory. The biological activity reduces at >40–45°C as well, but overheating is usually not an issue.

Figure CS1-24: Relative digestion rate as a function of temperature

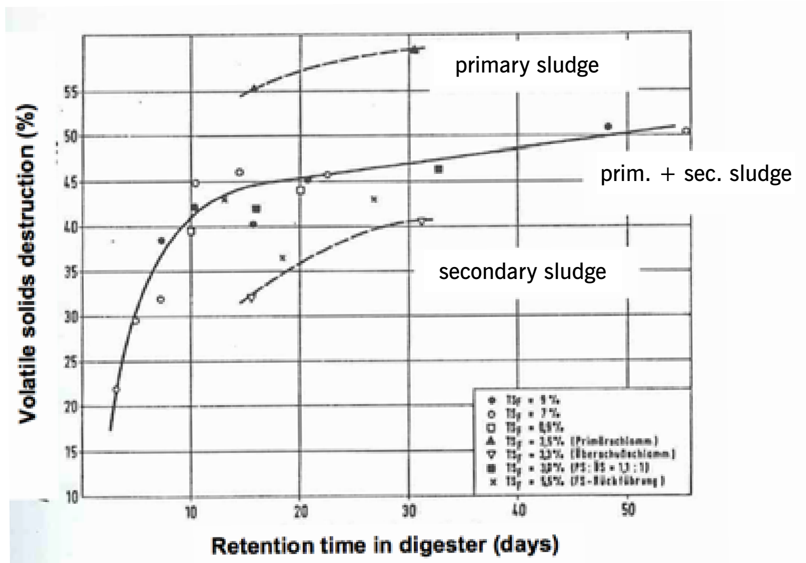


Source: Haandel and Lettinga 1994.

Table CS1-5: Biogas systems combined with anaerobic digesters: General design parameters and key characteristics

BIOGAS DESIGN	*Volatile solids (VS) concentration of sludge:* (a) primary sludge: average 75 percent (65–85 percent); (b) very high-loaded secondary sludge (sludge age ≈ 1 day): average 72 percent (65–80 percent); (c) secondary sludge of C elimination only (sludge age ≈ 5–10 days): average 70 percent (65–75 percent); (d) secondary sludge of N elimination (sludge age ≈ 10–15 days): average 68 percent (62–75 percent); (e) secondary sludge of extended aeration facility (sludge age > 20 days): average 65 percent (60–70 percent); (f) trickling filter sludge: average 70 percent (65–75 percent); (g) UASB sludge: average 55 percent (50–60 percent) (Bauerfeld et al. 2009; DWA 2003a; Buchauer 1996). *Volatile solids destruction:* (a) primary sludge: 55–60 percent; (b) secondary sludge: 30–40 percent; (c) mixture of primary and secondary sludge: 40–50 percent (Kapp 1984; Roediger et al. 1990).

Figure CS1-25: Volatile solids (VS) destruction over time



Source: Kapp 1984.

Note: Undigested primary or secondary sludge rarely has a VS content greater than 70 percent. The maximum VS destruction in a conventional mesophilic digester is about 50–60 percent, as indicated in figure CS1-25. Hence, digested sludge under these conditions usually does not achieve a VS content below 50 percent. Typically, digested sludge has a VS content between 50 and 55 percent. A VS content above 60 percent indicates a malfunction whose causes should be investigated.

Gas production: (a) primary sludge: 900–1000 L/kgVS destroyed; (b) secondary sludge: 700–800 L/kgVS destroyed; (c) mixture of primary and secondary sludge: 800–900 L/kgVS destroyed (Kapp 1984; Roediger et al. 1990).

Typical biogas characteristics: (a) methane CH_4: 60–70 percent; (b) calorific value: 6.0–7.0 kWh/Nm^3.

BIOGAS TREATMENT

Any biogas system should always contain the following:

- A foam trap (for removal of foam and particles from gas)
- Condensate removal (for removal of condensing gas humidity)

If necessary, one or several of the following technologies might also be used for further biogas cleaning. The main substances that require improved treatment usually are H_2S and siloxanes (Frey 2012; DWA 2011; WERF 2010b):

- Biological oxidation of H_2S in digester (= micro-aeration of digester)
- Precipitation of H_2S by Fe dosage into digester
- Adsorption reactors (Fe, activated carbon)
- Wet scrubbers

The first two options are the most common and economical solutions for H_2S removal in medium- to large-sized plants, and sometimes the processes are combined. In smaller facilities, adsorption in Fe filters prove easier to operate and more economical. Activated carbon is nowadays the standard solution for siloxane removal. Wet scrubbers are only applied in special cases, since this is typically the most costly, albeit one of the most efficient, technologies. Table CS1-6 summarizes typical features of biogas treatment.

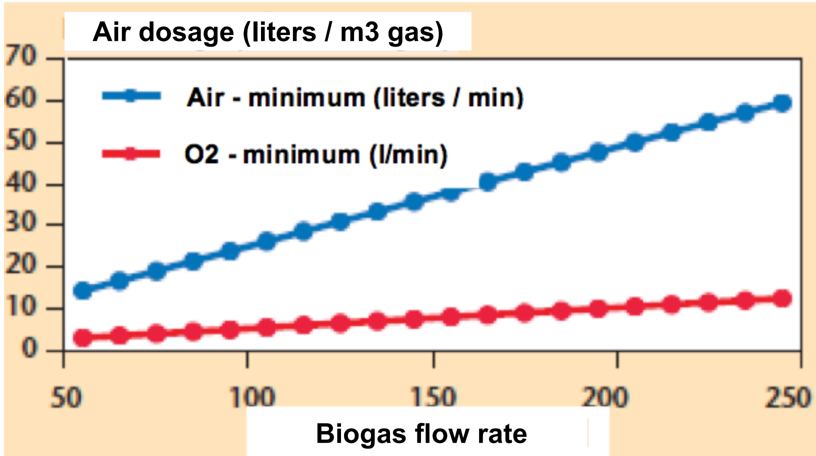
Table CS1-6: Biogas treatment: typical parameters, requirements, implications

Parameter	Typical value in biogas*	Typical requirement for CHP**	Implications
Water	n.a.	<60–85 percent humidity	Corrosion
H_2S (mg/m^3)	434	<200–450 percent	Corrosion, scaling, shorter intervals for lubrication oil and spark plug changes
Total siloxanes	15	<2–6	Scaling and wear and tear in combustion chamber and catalyzer, shorter maintenance intervals
Total chlorine and fluorine	434	<100	Problems with corrosion and catalyzer

***According to VSA (2012b), DWA (2011), and information from various manufacturers.*
n.a. = not available.

Micro-aeration: The air supplied into the digester should be introduced at several points just above the liquid level. This is where the desulphurizing microorganisms that utilize the oxygen have optimal conditions for their proliferation (Prochazka et al. 2013; Jäkel 2007). The required air quantity depends on the amount of H_2S that should be oxidized. Figure CS1-26 presents the necessary dosages for aeration. Mercato-Romain and others (2013) also report on the low air supplies needed for efficient H_2S removal.

Figure CS1-26: Minimum air dosage for H_2S removal by micro-aeration in case of gas with 2000 ppm H_2S

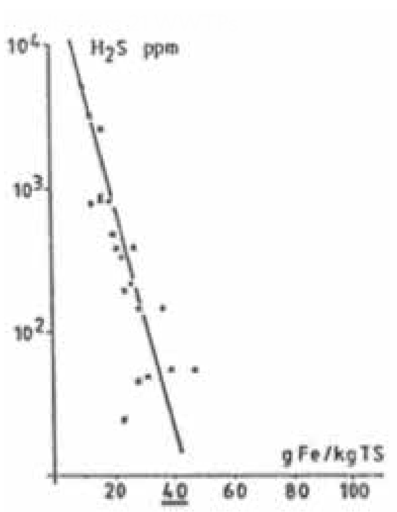


Source: Jäkel 2007.

Precipitation of H_2S by Fe dosage into digester: The required Fe dosage rate can be determined in two ways: by stoichiometric considerations or by introducing Fe so as to achieve a certain Fe/DS ratio in the digester.

- Stoichiometrically needed are 1.75 g Fe/g S-eliminated. By using a typical overdosage of 100 percent (Procházka et al. 2013; DWA 1996), one can calculate the necessary dosage of Fe.
- For safeguarding a typical H_2S <200 ppm in the treated gas, a concentration of 30–40 g Fe/kg DS is usually necessary (Ries et al. 1992); see figure CS1-27.

Figure CS1-27: H_2S concentration in biogas as function of iron content in digester sludge,



Source: Ries et al. 1992.

Adsorption to activated carbon (AC): Activated carbon cannot be designated for the removal of a specific substance alone. Rather, anything that can be adsorbed will be adsorbed to AC. Prior testing is hence indispensable if its exact cost needs to be calculated. Since H_2S is frequently the dominant compound that is being adsorbed, as a rough first guide one can estimate the needed AC quantities according to a typical adsorption capacity of 0.2–0.5 kg S per kg AC.

GAS HOLDER

The gas holder is meant to balance biogas production with biogas utilization. The gas holder should not be confused with a "storage tank"; rather, it serves as a small "balancing tank."

The daily fluctuation of biogas production depends primarily on the digester's sludge-feeding regime. If feeding is done only once a day, biogas production peaks after about two to three hours. For balancing this single peak, a holder volume of about 15–20 percent of daily production is usually sufficient (VSA 2010). Based on that background, gas holders are typically designed for a volume of about 10–30 percent of daily biogas production. The lower values prevail for large WWTPs; the higher values prevail for small WWTPs. But larger volumes of 50–80 percent are also sometimes recommended. However, whether the small amount of extra energy production facilitated by larger holder volume does, indeed, compensate for the substantial extra CAPEX is questionable.

Technically, low-pressure gas holders are the typical standard today. Two main types dominate the market:

- *Double membrane biogas holder:* This type of gas holder features two membranes: an outer membrane that is more robust and meant to protect against atmospheric conditions (sun, wind, rain, snow) and an inner membrane that contains the biogas. The space between these two membranes is connected to a blower that regulates the gas storage pressure. The higher the pressure, the better is the gas holder's resistance against wind, but its energy consumption also increases. The typical compromise is a pressure range of some 20–80 mbar. The membrane material is usually a PVC-coated polyester fabric. Double membrane gas holders can be installed as standalones on the ground or on top of the digester.

- *Single membrane biogas bags:* Gas bags are installed inside buildings or in specially constructed containers. Their shape can be easily adjusted to requirements. Usually they operate without pressure.

Figure CS1-28 shows two typical examples of biogas holders.

Figure CS1-28: Double membrane gas holder and biogas bag

Sources: Arrudas WWTP, Brazil (double membrane gas holder); Sattler, www.sattler-ag.com (biogas bag).

FLARE

A flare is an indispensable element of any WWTP with biogas production. It should be considered a safety installation, rather than a standard operating unit. The shorter its operation time, the higher the percentage of biogas utilization, and, thus, the greater the financial benefits.

To fulfil its safety function, the flare must be able to cope with the maximum possible hourly biogas production rate. Underdesign is to be avoided. An example is depicted in figure CS1-29.

Figure CS1-29: Gas flare

Source: Onca WWTP, Brazil.

BIOGAS UTILIZATION

Options for biogas utilization include the following (WERF 2012a; Frey 2012; DWA 2011):

- Burner
- Fuel cells
- Stirling motor
- Direct drive engines
- Supply into natural biogas supply systems
- Utilization as fuel for vehicles
- Co-generation
- Microturbines

Burners only produce thermal energy. This form of energy is usually of substantially lower economic value than electric power, and it is rarely required in East Asia and other warm regions.

Fuel cells have not yet reached a technical state of the art that would make their application easily possible anywhere. Even though they achieve relatively high electric efficiencies of >40 percent, they require extensive biogas treatment, they pose increased safety risks (H_2), and their CAPEX is still too high for viable application. For instance, the U.S. EPA indicates that fuel cells will only be economically viable if prices decline (U.S. EPA 2006), and WERF (2010b) describes them as one of the most expensive CHP technologies. Microbial fuel cells, a novel development, require further research and have not yet reached the point where they could be recommended for wider application (Rulkens 2007; Kletke et al. 2010; WERF 2011b).

The *Stirling motor* has very low electric efficiency of just slightly above 20 percent. It requires extensive maintenance of pistons and is only available in the market in very small units of about 10 kW.

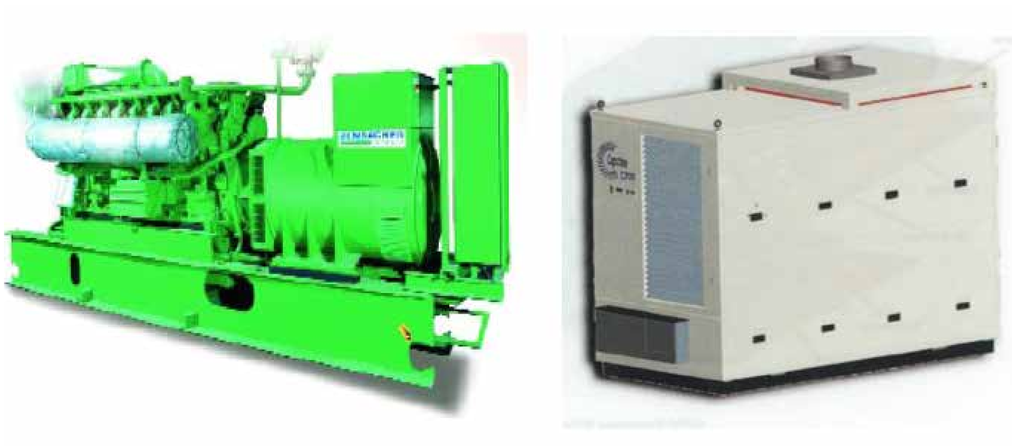
Direct drive engines may be considered for powering large consumers of electric energy, such as major pumps or aeration blowers. Even though it is sometimes stated that direct drive technology is cheaper and more efficient than co-generation or microturbines (Monteith et al. 2006), usually there are no substantial cost differences, and efficiencies are not, indeed, superior. Thus, even though they represent a robust technology, direct drive applications are not widely used.

Supply into natural biogas systems requires, as a matter of fact, a nearby biogas pipeline, which is frequently not available. Should this condition be met, it still faces a series of additional challenges: (a) technical: increase of CH_4 content, very strict criteria for H_2S and other quality parameters, pressure increase, need for constant supply; (b) legal; and (c) financial.

Utilization as fuel for vehicles has been applied in various locations worldwide. However, when considering this option, several issues have to be resolved: compliance with normed fuel quality requirements, safety aspects, and, not least, the storage issue due to constant supply but only intermittent utilization.

Hence, what is typically left as the most economical option for biogas reuse at WWTPs is combined production of electricity and heat (CHP) through *co-generation* or *microturbines*. Figure CS1-30 presents typical examples.

Figure CS1-30: Co-generation and microturbine



Sources: GE Jenbacher (co-generation); Capstone (microturbine).

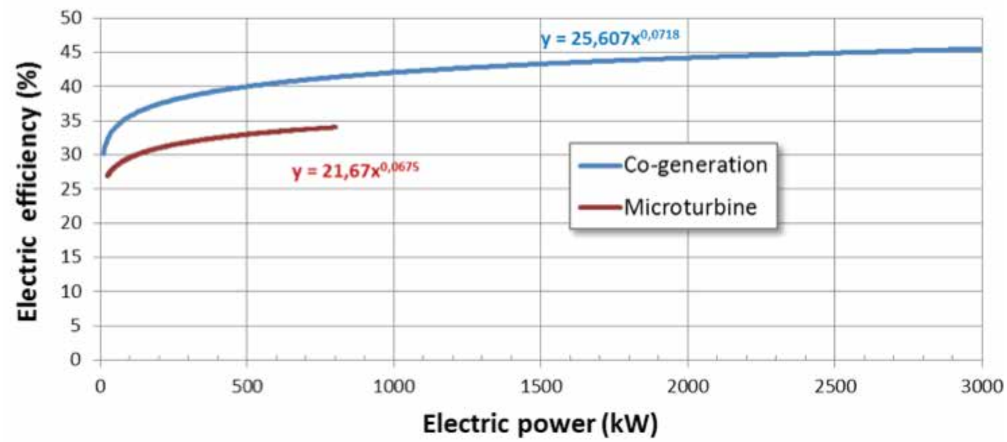
In co-generation, the biogas is burned in a motor, and a generator converts the mechanical energy into electric energy. In a microturbine, the biogas is also burned, but at substantially lower temperatures and at a surplus of oxygen. The burned gases move a turbine, from which a generator then produces the electric energy. The surplus of oxygen results in lower gas emissions (for example, less NO_x, and CO), and the turbine shows less wear and tear than a motor.

Co-generation is available on the market with electric unit capacities from some 30 kW up to several thousand kW. The maximum electric capacities of microturbines are lower, currently ranging from 30 kW to 600 kW per unit.

Figure CS1-31 presents typical electric efficiencies of these installations. They reflect average characteristics from products of numerous manufacturers available on the market.

In CHP, about 50 percent of the biogas's calorific value is converted into thermal energy, and about 10 percent is lost. No extra fuel is needed for the installations' operation.

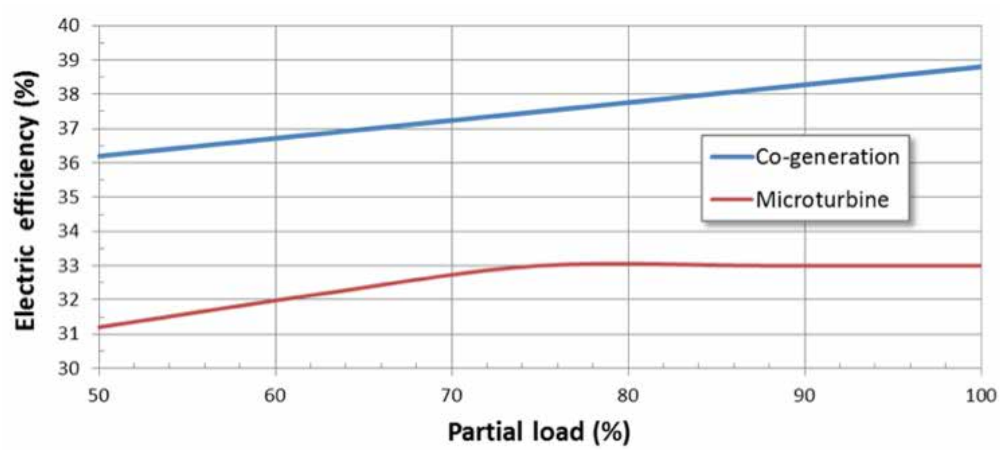
Figure CS1-31: Average electric efficiencies of co-generation and microturbine at full load



Sources: Values based on ASUE 2011 (for co-generation); VSA 2012b (for microturbines).

Electric efficiencies at partial load, presented in figure CS1-32, only serve to demonstrate the different characteristics of co-generation and microturbines: while co-generation immediately loses efficiency under partial load, microturbines have a near-constant efficiency for awhile, and only then does the efficiency decline.

Figure CS1-32: Exemplary electric efficiencies of co-generation and microturbine at partial load



Source: VSA 2012b.

While co-generation is a well-known system, microturbines are not yet generally known, but they could offer substantial advantages in East Asian countries due to their simpler technology and lower maintenance needs. Their requirements for gas cleaning are less strict; after all, this system was originally developed for military purposes with a focus on robustness. Their main disadvantages are slightly higher CAPEX and lower electric efficiency of about 30 percent. The key differences between co-generation and microturbines are summarized in table CS1-7.

Table CS1-7: Co-generation versus microturbines

Criteria for comparison	Co-generation	Microturbine
Electric efficiency	≈30–40 percent	≈30 percent
Electric efficiency reduction at partial load	Marked reduction	Less marked reduction
Thermal efficiency	≈50 percent	≈50 percent
Requirement toward gas quality	High	Medium-high
Exhaust emissions to open air	Higher emissions	Lower emissions
Noise emissions	High	Medium
CAPEX	Lower	Higher
OPEX	Higher	Lower

Sources: VSA 2012b; Frey 2012; WERF 2010b; VSA 2010; Geyer and Lengyel 2008.

CASE STUDY
2: TRICKLING FILTER + SLUDGE DIGESTION

2.1. BACKGROUND, PROCESS DESCRIPTION

2.1.1. Data sources

Trickling filters (TF) are an "old" technology that has been in use for many decades. In developed countries, TFs used to be more widespread, but nowadays the total number of TFs has fallen below 10 percent of the total number of WWTPs (see, for example, figure I-8 in the main body of the report). The main reason TF technology was pushed back is its low flexibility for influencing treatment efficiencies. An operator can influence the efficiency of TFs in some ways, but not to the extent possible with CAS.

Two key aspects stand out: the TF's simplicity of operation and its low energy consumption. In a CAS plant, approximately 60 percent of energy consumption is required for aeration of bioreactors (compare with case study 1); this energy item is eliminated completely in TFs that use natural air draft instead. If this reduction in energy consumption is combined with mesophilic digestion and biogas utilization, the goal of an energy-independent WWTP can become realistic. This feature, combined with simple operation (which in many environments is a big advantage in itself), is particularly attractive.

For case study 2, therefore, an example was selected of a large modern TF plant in a country with a subtropical climate: the main WWTP for Nicaragua's capital, Managua, in Central America. This facility is designed for about 1.1 million capita and was commissioned in 2009. It has several interesting features, such as the use of lamella settling tanks (both for PST and SST) to reduce the plant's footprint, anaerobic digesters operating at ambient temperature, and solar sludge drying. The municipal wastewater in Managua is mainly domestic; hence, there is no interference from industrial discharges that require special attention. The WWTP has been in operation for five years now, and all conclusions about it can be based upon reliable data. Biogas has so far been collected and metered but is not yet utilized. Now that the biogas production yield is established, a biogas utilization project is in the pipeline. The plant is owned by ENACAL (Empresa Nicaragüense de Acueductos y Alcantarillados), and the first five years of operation from which the data have been drawn for this case study were contracted to Biwater International Ltd.

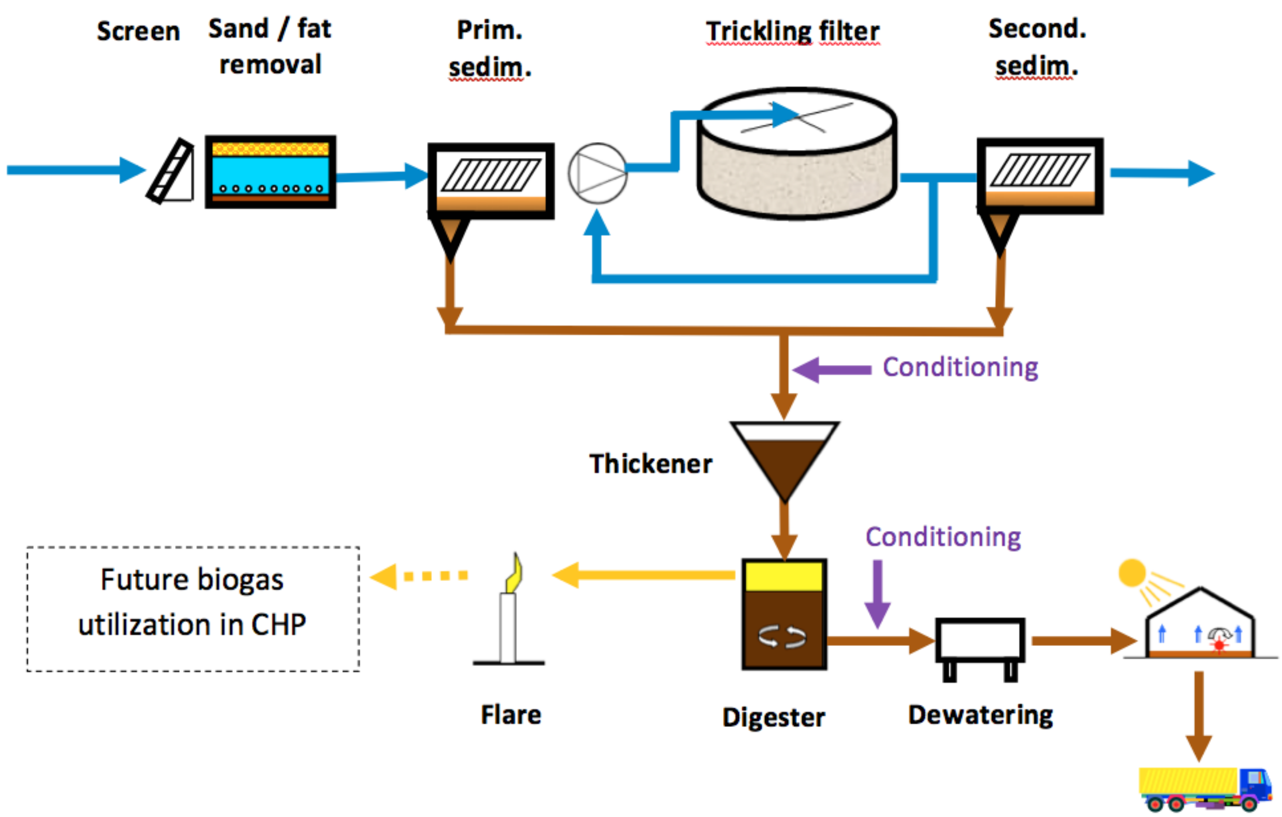
2.1.2. Wastewater management

The TF technology applied in Managua is a classical concept (see figures CS2-1 and CS2-2): preliminary treatment (screen, sand/fat removal) followed by settling in the primary sedimentation tank (PST) and subsequent pumping onto the TFs. The treated wastewater and sludge flushed out from the TFs settle in the secondary sedimentation tanks (SSTs). The effluent is subsequently discharged to the nearby Managua Lake. Part of the treated wastewater is recirculated to the TF pumping station to ensure a certain minimum wetting rate of the filters (10–25 mm/pass) during periods when the influent flow rate is insufficient for that purpose. The TFs themselves have a filter depth of 5.4 meters and are filled with cross-flow plastic media with a specific surface area of 100 m^2/m^3.

Both PSTs and SSTs are equipped with lamella plates to decrease their footprint. This was mainly done to minimize the size and cost of a flood protection dam that surrounds the whole plant. The design of the lamella settlers is based on conventional criteria.

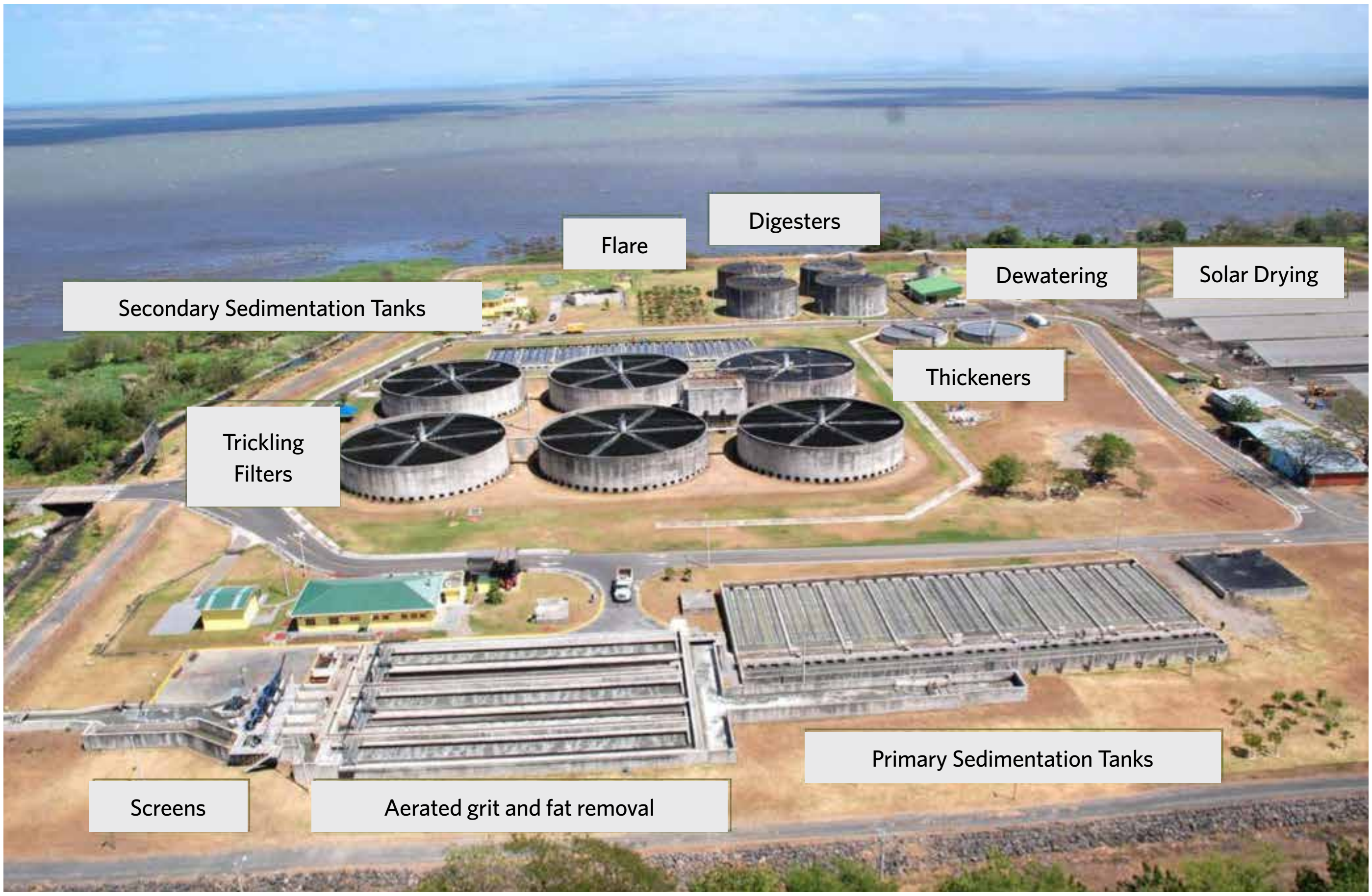
For details of actual wastewater characteristics and other parameters, see tables CS2-1 and CS2-2.

Figure CS2-1: Simplified flow scheme of TF system and sludge treatment at Managua WWTP



Source: Authors.

Figure CS2-2: Aerial view of Managua WWTP



Source: ENACAL—Managua WWTP.

2.1.3. Sludge management

The sludge from PSTs and SSTs is combined and thickened by gravity to about 3 percent DS. For improved separation of solid and liquid phases at the high temperatures in Managua, some polymers are added prior to thickening. The sludge is then digested in four closed digesters at ambient temperatures without any heating (the annual average air temperature is ≈27.5°C, with monthly averages over the year between 26° and 29°C). The digesters are the shallow cylindrical type (see table CS1-22), with a ratio of diameter to sludge depth of about 2.2:1.0. They are constructed out of concrete without thermal insulation. Each digester is equipped with three internal draft mixers. Digested sludge is dewatered in belt filter presses to about 30 percent DS and further dried in a solar drying facility to >90 percent DS. The final product goes into agriculture/sanitary landfill. Meyer-Scharenberg and Pöppke (2010) describe the sludge treatment in more detail.

A project is in the pipeline to install additional mechanical thickeners to further increase the DS content of the sludge prior to digestion, up to about 6 percent DS. This will halve the sludge volume to digest and double the retention time in the digesters.

For details of actual sludge characteristics, see table CS2-2. For details on general design parameters and key characteristics of anaerobic digesters, see table CS1-4.

2.1.4. Energy management

The generated biogas is currently only flared. This was done in previous years to collect actual operation data regarding biogas generation in the unheated digesters because the forecast of biogas production was deemed unreliable in the design stage due to the uncertainties surrounding the lack of heating. The biogas assessment has finally taken place for the years 2011–12 (FWT 2013b). Consequently, the biogas utilization project is in the pipeline now, and its elements are already defined as follows:

- Foam trap
- Condensate removal
- Optional: activated carbon filter for H_2S and siloxane removal
- Gas holder (1000 m3)
- New flare
- CHP: five microturbines with 200 kW each (total electric power = 1,000 kW)
- Pipes, valves, and so on, as required

For details of actual biogas characteristics, see table CS2-2. For details on general design parameters and key characteristics of biogas systems combined with anaerobic digesters, see table CS1-5.

2.2. Analysis

2.2.1. Wastewater influent, effluent, and other parameters of interest

The data presented in tables CS2-1 and CS2-2 are taken from an assessment that took place for the two years between January 2011 and November 2012 (FWT 2013b).

Table CS2-1: Actual influent and effluent data of Managua WWTP, average data from 1/2011 to 11/2012

			Managua WWTP
Number of WWTPs			1
Pop. equivalents	avg. actual	PE_{60}	447,000
	max. actual	PE_{60}	606,000
WASTEWATER QUANTITY			
Specific wastewater production		$m^3/PE_{60}/y$	82
		$L/PE_{60}/d$	225
WASTEWATER QUALITY			
COD	Influent	mg/L	505
	Effluent	mg/L	101
	Elimination	%	80
BOD_5	Influent	mg/L	248
	Effluent	mg/L	28
	Elimination	%	89
TSS	Influent	mg/L	259
	Effluent	mg/L	31
	Elimination	%	88
N_{total}	Influent	mg/L	27.6
	Effluent	mg/L	17.5
	Elimination	%	37
NH_4-N	Effluent	mg/L	n.a.
NO_3-N	Effluent	mg/L	n.a.
P_{total}	Influent	mg/L	3.7
	Effluent	mg/L	1.7
	Elimination	%	54

Source: FWT 2013b.

Notes: 1 cap = 46.5 g BOD_5/d in Nicaragua; 1 PE_{60} = 1.29 $PE_{46.5}$ = 1.29 cap in Nicaragua; m3/PE_{60}/y x 1.29 = m3/cap/y in Nicaragua; L/PE_{60}/d x 1.29 = L/cap/d in Nicaragua.

n.a = not available.

Table CS2-2: Key characteristics of Managua WWTP

			Actual	Design
GENERAL				
Daily flow rate	average	m3/d	100,750	182,563
	95 percentile		142,572	297,302
BOD_5 load	average	kg/d	26,846	50,663
COD load	average	kg/d	55,469	101,326
TSS load	average	kg/d	28,178	30,398
BOD_5 effluent	average	mg/L	28	—
	90 percentile		42	90
COD effluent	average	mg/L	101	—
	90 percentile		147	180
TSS effluent	average	mg/L	31	—
	90 percentile		50	80
WASTEWATER TRAIN				
PST retention time	average	h	1.37	—
	5 percentile		0.79	0.50
TF volumetric load	average	$gBOD_5/m3/d$	0.43	1.46
FST surface load	average	m/h	0.70	—
	5 percentile		0.93	1.48
SLUDGE TRAIN				
Thickened sludge	average	m3/d	800	795
		kgDS/d	22,600	39,735
VS of raw sludge	average	%	67	75
Digester retention time	average	d	22.7	20.0
	5 percentile		15.0	
VS destruction in digester	average	%	49	50
VS of digested sludge	average	%	51	60
BIOGAS TRAIN				
Biogas production	average	m3/d	7,159	13,038
	95 percentile		9,466	
Specific biogas product.	average	$L / kgVS_{destroyed}$	950	875
Calorific value biogas	average	kWh/m3	8.4	—
H_2S	average	ppm	130	—
	max.	ppm	210	—
Air temperature	average	°C	29.5	—

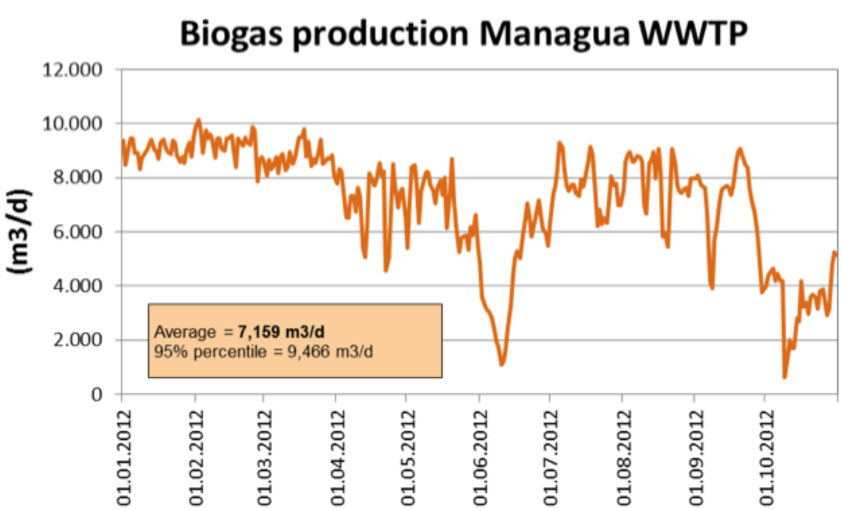
Note: Most of the above actual data refer to the period January 2011 to November 2012. The biogas data refer to January 2012 to October 2012.

2.2.2. Biogas production and potential for energy generation

Biogas production

Figure CS2-3 shows biogas production in the digesters during the first ten months of 2012, which is mostly constant. Just during two periods, in June and October respectively, is there a pronounced reduction in biogas production to almost zero. These reductions were caused by strongly reduced influent flow rates to the WWTP due to problems with the influent pumping station. Under normal operating conditions, these fluctuations need not be expected, and "normal," undisturbed biogas production should be about 5–10 percent higher than the metered average during this period. Hence, if there had been no problems with influent pumping, the documented average specific biogas production rate of 950 $L/kgVS_{destroyed}$ over the whole period would have easily reached about 1000 $L/kgVS_{destroyed}$. This is on the upper end of the biogas production range indicated in table CS1-5. Thus, case study 2 demonstrates that even unheated digesters can be very efficient in warm climate environments.

Figure CS2-3: Daily biogas production at Managua WWTP



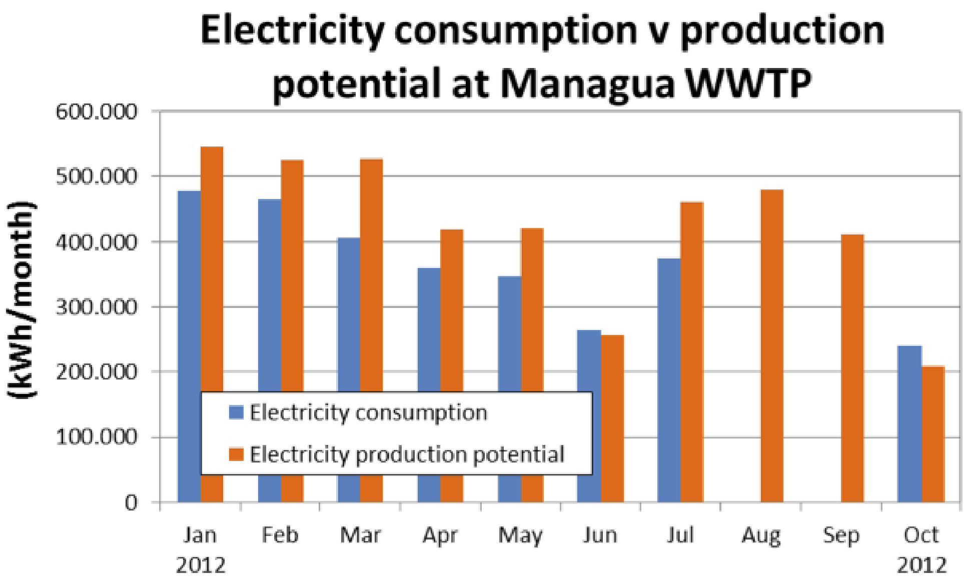
Source: FWT 2013b.

Potential for energy generation

The calorific value of the biogas was determined by the Laboratorio de Geoquímica Geotérmica of the Ministerio de Energía y Minas, Managua, Nicaragua. It found an average value of 8.4 kWh/m^3. Since this value is surprisingly high, a more conventional value of 6.5 kWh/m^3 was chosen for the analysis presented below (see table CS1-5). Thus, it is possible that the actual potential of energy generation is as much as about 25 percent higher than calculated. The electric efficiency of the proposed five microturbines with 200 kW each is assumed as 30 percent (see figure CS1-31).

The potential for electricity generation is calculated and compared to the actual total electricity consumption of Managua WWTP (figure CS2-4). The forecast average electricity production equals 426 MWh/month, which equals 5,110 MWh/year. Additionally, a potential for thermal heat generation from CHP was calculated as 781 MWh/month, which equals 9,370 MWh/year (FWT 2013b).

Figure CS2-4: Power generation potential of actual biogas production, compared to actual electricity consumption at Managua WWTP



Source: FWT 2013b.

Note: No electricity consumption data were available for the months of August and September 2012.

From the results, the following conclusions can be derived:

- Typically, the biogas' electricity production potential is 10–20 percent higher than the actual electricity consumption, even though in this case study less efficient microturbines are planned instead of more efficient co-generation.
- Microturbines are preferred over co-generation in this case for the following reasons: high efficiency at partial load, somewhat lower requirements toward gas quality, less emissions to air, less noise, longer maintenance service intervals, and, most important, lower OPEX (see also table CS1-7).
- Only during two months (June and October) when there were problems with the influent pumping stations was the consumption slightly higher than the electricity potential.
- The average electricity consumption equals 9–10 kWh/PE_{60}/year at Managua WWTP. This is substantially lower than comparable values for CAS. On an annual basis, a WWTP based on TF technology and sludge digestion like that in Managua can be operated with a positive electric energy balance. The electricity supply from the public grid serves only as "safety net" for brief peak periods and during operational problems.

Table CS2-3: Biogas and power generation potential of TF + digester at Managua WWTP

	Retention time in PST (h) 1.4
Biogas production	
- N elimination (L/PE_{60}/d)	—
- C elimination (L/PE_{60}/d)	16.0
Electric efficiency CHP (%)	30
Thermal efficiency CHP (%)	55
Calorific value of biogas (kWh/m^3)	6.5
Power generation	
- N elimination (kWh/PE_{60}/year)	—
- C elimination (kWh/PE_{60}/year)	11.4
Thermal energy generation	
- N elimination (kWh/PE_{60}/year)	—
- C elimination (kWh/PE_{60}/year)	20.9
Total energy generation	
- N elimination (kWh/PE_{60}/year)	—
- C elimination (kWh/PE_{60}/year)	32.3

Source: Authors' calculation.

Note: L/PE_{60}/d x 16.67 = L/kg BOD_5/d; kWh/PE_{60}/y x 16.67 = kWh/kg BOD_5/y.

When comparing the above results with international experiences, the emerging picture might be confusing at first glance, and hence requires a closer look:

- A basic "problem" is that most documented energy data on TFs originate from facilities in developed countries, and very few are published about TFs in warm climate zones. The TFs in developed countries are adjusted to their local requirements, which usually call for C+N elimination in (at least seasonally) cold climates. This usually implies additional electric power consumption for forced ventilation; for additional denitrification stages; for higher recirculation rates; or for separate installations for enhanced P removal. Figures I-6 and I-7 in the main body of the report show European and U.S. energy consumption results for TFs of 25 kWh/PE_{60}/y and 30–40 kWh/PE_{60}/y on average, respectively. These average energy consumption values are still about 25 percent lower than for CAS, but the differences from CAS are not very significant.
- Second, there are more efforts underway to optimize CAS plants from the energy utilization point of view, whereas TF plants are rarely subject to the same efforts. As TFs are classified as "low-energy technology," operators usually think their energy-saving potential is low. Only a few large-scale examples of energy optimization efforts at TFs are reported in the literature. If optimization is carried

out, nonetheless, the gap between TF and CAS widens. For instance, one recent study by Witzgall and others (2013) found that an optimized TF for C+N elimination (in the United States) only consumes half the electric power of a comparable CAS system.

- Third, case study 2 demonstrates that energy numbers from Europe and the United States should not be uncritically applied to TFs in warm climates. Rather, when there is a requirement for C elimination only in warm climates, TFs can apparently operate with an electricity consumption of <10 kWh/PE_{60}/y, including sludge digestion and even sludge drying. This not only lowers the electricity bill in general, but it offers an even more attractive scenario: these facilities are indeed capable of fully covering their own power needs through autogeneration of electric energy from sludge at the WWTP.
- Fourth, it is important to understand that a nitrification requirement does not fundamentally change the electricity requirement of TFs. Nitrification only implies a need for larger TFs, but the pumping head always remains the same.

2.2.3. Operation capacity needs, biogas safety

For digester operation the same principles always apply, independent of digester location. All the relevant operation and safety issues for digesters have already been discussed in much detail in case study 1 (see section 1.2.3).

2.2.4. Institutional aspects, energy costs

The clear preference in Managua is for utilization of generated electricity onsite. But the numbers indicate there could be small temporary power surpluses that cannot be economically balanced in gas holders or batteries. Hence, the question of electricity supply to the public grid is not a dominant issue, but it is not irrelevant, either.

Nicaragua has no legal restrictions that would impede the supply of electric energy into the public grid. Since this approach would be novel to the country, however, no clear price has yet been set. This price would depend on the result of negotiations between the owner of the WWTP (ENACAL) and the public supplier of electric energy (Gas Natural).

ENACAL currently pays a low tariff of US$0.08/kWh for electricity purchased from the public grid. The conventional commercial tariff stands at US$0.32/kWh.

FWT (2013b) therefore based all financial assessments of investment in biogas utilization on three different electricity cost scenarios: US$0.08, US$0.13, and US$0.3/kWh, respectively. For results, see section 2.2.8.

2.2.5. GHG reduction and CDM co-financing

General

For general information on CDM co-financing and the price of carbon credits, see section 1.2.5.

Specifics of case study 2

As pointed out in section 1.2.5, due to the "additionality" criterion, in the new construction of a digester completed with CHP the energy production component under CDM should be considered, but not methane elimination, since methane is created and eliminated by the project.

Nicaragua's specific energy mixture causes the GHG emissions from electricity generation shown in table CS2-4.

With a forecast annual electricity production potential from biogas of 5,110 MWh/year, the potential **GHG reduction is 2,350 tons CO_{2e}/year** for case study 2 at present, based on Nicaragua's average energy mixture.

Table CS2-4: GHG emissions per kWh for electricity generation in Nicaragua, as compared to the world

REGION	1990 gCO/kWh *	2010 gCO/kWh *
Nicaragua	345	**460**
World	586	**565**

Source: IEA 2012.

Note: The table shows CO_2 emissions from fossil fuels consumed for electricity generation, in both electricity-only and combined heat and power plants, divided by output of electricity generated from fossil fuels, nuclear, hydro (excluding pumped storage), geothermal, solar, wind, tide, wave, ocean, and biofuels. Both main activity producers and autoproducers have been included in the calculation.

2.2.6. CAPEX structure

CAPEX requirements for a biogas utilization project at Managua WWTP have been estimated by FWT (2013b). All components were based on actual financial bids from potential suppliers. The outcome (table CS2-5) can be considered quite realistic and reflects the local specifics prevailing in Nicaragua.

Table CS2-5: CAPEX for biogas utilization project at Managua WWTP

	CAPEX	
	US$	EUR
Biogas pretreatment, gas holder, flare	877,500	650,000
CHP with microturbines	1,431,000	1.060,000
SUBTOTAL	2,308,500	1,710,000
Contingencies 10 percent	230,850	171,000
Consulting 10 percent	230,850	171,000
Price adjustment 5 percent	115,425	85,500
TOTAL	**2,885,625**	**2,137,500**

Source: FWT 2013b.

The additional cost for an optional biogas treatment to remove H_2S and siloxanes was estimated at US$1 million. For the time being, however, this was not considered necessary, given the unproblematic results of the gas analysis available to date.

Note that the above CAPEX will be sufficient to cover the biogas project needs up to the year 2025. It assumes an increase in influent load (and thus in biogas production) of about 55 percent due to large increases in the sewer connection rate between 2013 and 2025. The influent pollution load to Managua WWTP will thus increase to ≈700,000 PE_{60} on annual average and to ≈950,000 PE_{60} during peak periods.

2.2.7. OPEX structure

OPEX requirements for a biogas utilization project at Managua WWTP were estimated by FWT (2013b); see table CS2-6. All components were adjusted to the local specifics prevailing in Nicaragua.

Table CS2-6: OPEX for biogas utilization project at Managua WWTP

	OPEX 2012		OPEX 2025	
	US$/y	EUR/y	US$/y	EUR/y
Biogas pretreatment, gas holder, flare	37,800	28,000	37,800	28,000
CHP with microturbines	90,450	67,000	90,450	67,000
SUBTOTAL	128,250	95,000	128,250	95,000
Saving in case of electr. tariff = 1.86 C$/kWh	-278,100	-206,000	-483,300	-358,000
TOTAL in case of electr. tariff = 1.86 C$/kWh	**-149,850**	**-111,000**	**-355,050**	**-263,000**

Source: FWT 2013b.

Consequently, if electricity unit cost remains unchanged at C$1.86/kWh, OPEX will reduce by US$150,000/y (EUR111,000/y) in 2012 and by US$355,000/y (EUR263,000/y) in 2025.

2.2.8. Viability of investment in biogas utilization

Table CS2-7 summarizes several cost indicators from case study 2.

Table CS2-7: Cost indicators for biogas utilization project at Managua WWTP

		2012	2025
Average influent load	$PE_{60,avg}$	447,000	700,000
Peak influent load	$PE_{60,max}$	606,000	950,000
CAPEX	US$	2,885,625	—
OPEX	US$/y	-149,850	-355,050
specific CAPEX	US$/$PE_{60,avg}$	6.46	4.12
	US$/$PE_{60,max}$	4.76	3.04
specific OPEX	US$/$PE_{60,avg}$	-0.34	-0.51
	US$/$PE_{60,max}$	-0.25	-0.37

Source: FWT 2013b and authors' calculation.

Note: 1 PE_{60} = 1.29 $PE_{46.5}$ = 1.29 cap in Nicaragua; US$/$PE_{60}$ x 16.67 = US$/kg BOD_5.

The specific CAPEX related to the future peak influent pollution in 2025 is thus US$3.04/$PE_{60,max}$ (EUR2.25/$PE_{60,max}$). The specific CAPEX related to the future average influent pollution in 2025 is thus US$4.12/$PE_{60,avg}$ (EUR3.05/$PE_{60,avg}$).

Relating the minimum OPEX savings to average annual influent pollution leads to savings of US$0.34/$PE_{60,avg}$ (EUR0.25/$PE_{60,avg}$) and US$0.51/$PE_{60,avg}$ (EUR0.38/$PE_{60,avg}$) in 2012 and 2025, respectively. It was concluded in FWT (2013b) that the project is financially viable. Total annual cost (CAPEX + OPEX) of Managua WWTP is expected to decrease with the new biogas utilization project.

2.3. Conclusions for TF + sludge digestion in East Asian countries

The key characteristics of sludge digestion, VS destruction, and biogas production, as already described in case study 1, remain unchanged, as these processes do not depend on location, but on the environmental conditions inside the digester. The reported biogas yield from the digesters in case study 2 confirms that unheated digesters in warm climate countries can achieve comparable results to heated digesters in cold climates. Indirectly, this result also demonstrates that the biogas potentials from CAS sludge and TF sludge are rather similar.

Case study 2 also confirms the feasibility of unheated sludge digesters in warm climates.

Compared to Nicaragua, the characteristics and quantities of sludge that could be expected from a TF plant in EAP are different. This change will become particularly evident where wastewater dilution is high and where septic tanks continue to be used in large numbers. In EAP, reduced quantities of primary sludge from PSTs and reduced sludge quantities in general are expected. This, in turn, means a shift toward more

TF sludge and less PS, resulting in reduced biogas potential.

Another possible change relates to wastewater treatment requirements. The TF in Nicaragua is designed for carbon removal only. If in EAP additional nitrification is required, CAPEX, of course, will increase due to a need for larger/more TF reactor volume, but OPEX will remain almost unchanged, since the same flow rate continues to be pumped to the same water head, and pumping is the dominant energy consumer. Only if denitrification is required, the energy requirements for additional mixing and recirculation will increase overall energy consumption. This might eventually lead to a situation where, with nitrification and denitrification, less than 100 percent of electricity consumption can be covered from biogas.

CAPEX could be similar in many cities in EAP and in Nicaragua. OPEX savings may be less in situations where wastewater dilution is high and sludge production is consequently lower, but they may also be higher where no dilution prevails and power unit cost is higher than in Nicaragua (US$0.08/kWh). In several EAP countries, electricity unit cost is higher than in Nicaragua. Thus, the financial value of the generated electricity is higher in these places as well. Consequently, the amortization periods of investment into sludge digestion and biogas projects in EAP need case-specific analyses.

Success stories in EAP

No TF plants in EAP with publicly accessible operation data are known to the authors of this technical note.

Since CAPEX levels in Nicaragua and EAP could be of similar magnitude, similar financial conclusions may be expected. The cost assessment, however, is also influenced by local wastewater specifics. Thus, in EAP, where the combined effects of wastewater dilution and the use of large numbers of septic tanks prevail, these projects can be expected to be less financially appealing than they are in Nicaragua. Such tentative assessments can lead to wrong conclusions, as shown by the application example of the assessment tool that was explained in the main body of this report. A minor difference in influent characteristics can trigger a completely different outcome with respect to financial viability. Therefore, a sound assessment of individual energy projects is highly recommended, particularly where electricity unit cost is high, where extra organic feedstock is available, and where safe sludge disposal becomes a matter of concern.

CASE STUDY 3: UASB

3.1. BACKGROUND, PROCESS DESCRIPTION

3.1.1. General background, data sources

Upflow anaerobic sludge blanket (UASB) technology was originally developed in the Netherlands. According to Haandel and Lettinga (1994), "The steep increase in energy prices in the 1970s reduced the attractiveness of aerobic treatment systems and intensified research efforts towards the development of systems with lower energy consumption." UASB reactors work well if a certain minimum wastewater temperature, ideally >20°C, is provided. Hence, the system is usually applied to industries with warm wastewater, and UASB reactors for municipal wastewater are de facto used exclusively in countries with warm climates. The first large-scale municipal UASBs were constructed in such places as Petregal (Brazil), Cali (Colombia), and Kanpur (India) in the 1980s. UASB is not yet widespread in the EAP countries focused on in this report, even though the climate there would be well suited. Apart from several large-scale Indian applications, a UASB pilot in Singapore has clearly demonstrated that this technology would work well in EAP (Cao 2011).

UASB technology digests both wastewater and sludge in the same reactor without a supply of oxygen. Hence, no separate sludge digesters are required, as they were in case studies 1 and 2, and electricity consumption is limited to wastewater pumping and supplementary installations, such as sludge dewatering. The anaerobic reactors are covered, and the biogas is collected. Since it is an anaerobic technology, which is usually not sufficient to meet required effluent standards, many UASB systems feature a subsequent aerobic polishing stage. This stage can use any aerobic technology, but the most common choice is ponds or trickling filter. CAS is only seldom used for that purpose.

Case study 3 is based on a comprehensive analysis of twenty-two UASB systems (FWT 2013c) in Minas Gerais, Brazil, that was conducted in the course of the project, "Despoluição da Bacia Hidrográfica do Rio Paraopeba." This project is co-financed by COPASA and KfW. All analyzed UASB systems are operated by COPASA (Companhia de Saneamento de Minas Gerais).

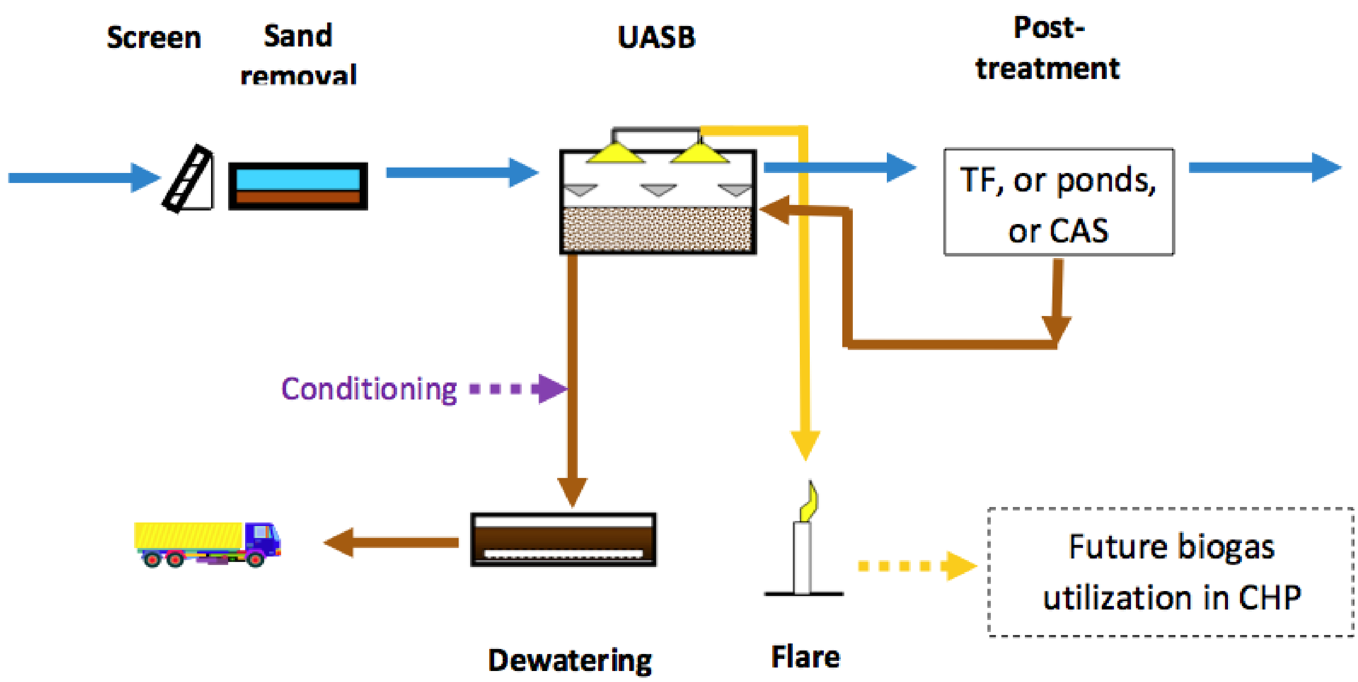
3.1.2. Wastewater management

The capacities of the investigated WWTPs cover a wide range, from about 10,000 PE_{60} to almost 1 million PE_{60}. Most operate within their capacity, and overloading is infrequent.

Preliminary treatment at all the plants is by means of screens and grit chambers. Specific fat removal is never available. A pumping station is always included, whether for influent pumping or pumping onto TFs.

The twenty-two investigated WWTPs polish their anaerobic effluents from the UASB as follows: twelve plants use trickling filters (TFs), four use ponds, one uses conventional activated sludge (CAS), and five have no polishing stage at all. Figure CS3-1 presents a simplified flow scheme for the WWTPs of case study 3, and figure CS3-2 shows an aerial view of one of the investigated plants. Table CS3-1 provides key characteristics of all the facilities.

Figure CS3-1: Simplified typical flow scheme of UASB systems for case study 3



Source: Authors.

Figure CS3-2: Aerial view on Onça WWTP, the largest investigated UASB plant (177 MLD)

Source: COPASA—Onça WWTP.

For more **details of actual wastewater characteristics** and other parameters, see section 3.2.1.

Table CS3-1: Key characteristics of UASB plants investigated for case study 3

WWTP	Technology	Capacity (m3/d)	Capacity (PE_{60})	Actual avg. annual load (PE_{60})
Curvelo	UASB-TF	8,747	54,300	41,050
Janaúba	UASB-ponds	3,854	20,157	25,280
Montes Claros_Vieiras	UASB-TF	42,738	320,533	128,100
Tres Marias	UASB-TF	3,065	20,673	16,183
Ipatinga_Rio Doce	UASB	32,704	165,385	206,640
João Pinheiro	UASB-TF	4,450	26,556	16,150
Lafaiete_Bananeiras	UASB-TF	7,486	49,081	18,617
Alfenas	UASB-TF	17,088	74,450	30,152
Caxambu	UASB-ponds	6,963	27,450	6,103
Itajubá	UASB	19,910	98,024	32,250
Lavras_Água Limpa	UASB-ponds	6,786	20,976	14,967
Lavras_Ribeirão Vermelho	UASB-ponds	13,515	59,383	29,521
Pouso Alegre	UASB	25,634	111,182	31,800
Varginha_São José	UASB	10,289	39,287	22,883
Varginha_Santana	UASB	18,985	79,990	29,367
Onça	UASB-TF	176,861	930,833	577,917
Betim Central	UASB-CAS	44,391	278,733	53,653
Nova Contagem	UASB-TF	7,442	44,683	12,667
Vale do Sereno	UASB-TF	2,354	10,210	9,263
Pará de Minas	UASB-TF	12,994	74,000	52,317
São José da Lapa	UASB-TF	3,228	20,600	7,450
Vespasiano	UASB-TF	3,944	33,341	10,837
Average		**21,519**	**116,356**	**62,417**
Total		**473,426**	**2,559,829**	**1,373,166**

Source: FWT 2013c.

Note: 1 cap = 54 g BOD_5/d in Brazil; 1 PE_{60} = 1.11 PE54 = 1.11 cap in Brazil.

3.1.3. Sludge management

At most of the WWTPs analyzed in case study 3, sludge is produced in two stages: (a) in the UASB reactors and (b) in the polishing stage. Sludge from the latter is generally conveyed back to the UASB for stabilization. Hence, typical sludge withdrawal is only from the UASB reactor. This sludge has an average DS content of about 3.8 percent and can be considered stabilized (VS = 56 percent on average). For that reason there is no particular need for further stabilization, and the sludge is just dewatered. Out of the twenty-two plants, seven have centrifuges for sludge dewatering, while the other fifteen use sludge drying beds.

For details of actual sludge characteristics and other parameters, see section 3.2.1.

General design parameters for UASB reactors according to state-of-the-art recommendations are compiled in table CS3-10. It is assumed that the interested reader is familiar with the background and microbiological principles of anaerobic digestion. The table summarizes selected key parameters for design and construction of UASB.

3.1.4. Energy management

In general, generated biogas is only flared at WWTPs. The biogas flow rate is only metered at a few of the WWTPs of case study 3, and, unfortunately, these few flow meters deliver implausible results. General state-of-the-art design parameters and recommendations for UASB biogas systems are compiled in table CS3-11.

3.2. Analysis

3.2.1. Wastewater influent and effluent and other parameters of interest

Table CS3-2 and figures CS3-3, CS3-4, and CS3-5 summarize the outcome of an assessment of the twenty-two UASB plants in case study 3 (FWT 2013c). The results cover the complete calendar year 2011, with data related to both wastewater and sludge characteristics presented.

Table CS3-2: Actual influent and effluent data of UASBs of case study 3, average data from 2011

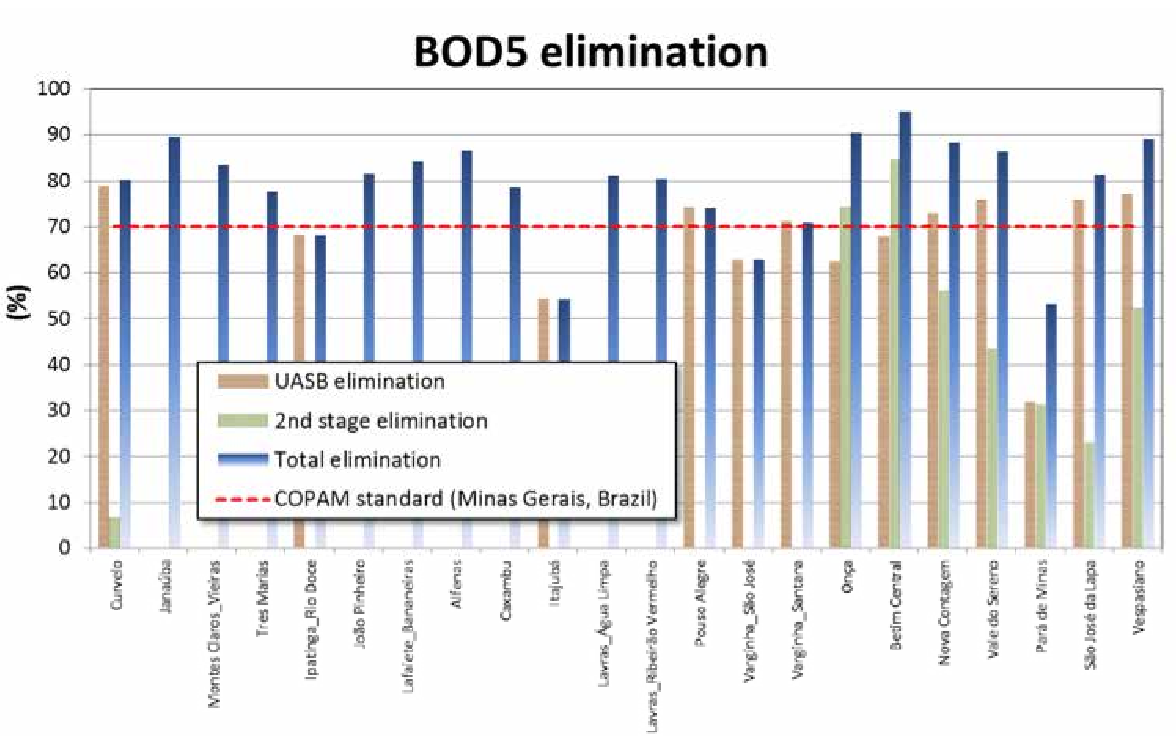
			UASBs
Number of WWTPs			22
Pop. equivalents	avg. actual	PE_{60} per UASB	62,417
	max. actual	PE_{60} per UASB	577,917
WASTEWATER QUANTITY			
Specific wastewater production		$m^3/PE_{60}/y$	80
		$L/PE_{60}/d$	220
WASTEWATER QUALITY			
COD	Influent	mg/L	697
	Effluent	mg/L	194
	Elimination	%	72
BOD_5	Influent	mg/L	297
	Effluent	mg/L	62
	Elimination	%	76
N_{total}	Influent	mg/L	n.a.
	Effluent	mg/L	41
	Elimination	%	n.a.
NH_4-N	Effluent	mg/L	38
NO_3-N	Effluent	mg/L	1
P_{total}	Influent	mg/L	7.1
	Effluent	mg/L	4.5
	Elimination	%	33

Source: FWT 2013c.

Notes: 1 cap = 54 g BOD_5/d in Brazil; 1 PE_{60} = 1.11 PE_{54} = 1.11 cap in Brazil; $m^3/PE_{60}/y$ x 1.11 = m^3/cap/y in Brazil; $L/PE_{60}/d$ x 1.11 = L/cap/d in Brazil. "Effluent" and "elimination" related to total WWTP (not UASB only).

n.a. = not available.

Figure CS3-3: BOD_5 removal efficiencies at WWTPs of case study 3



Source: FWT 2013c.

Table CS3-3: Key average characteristics of UASBs of case study 3, actual versus design

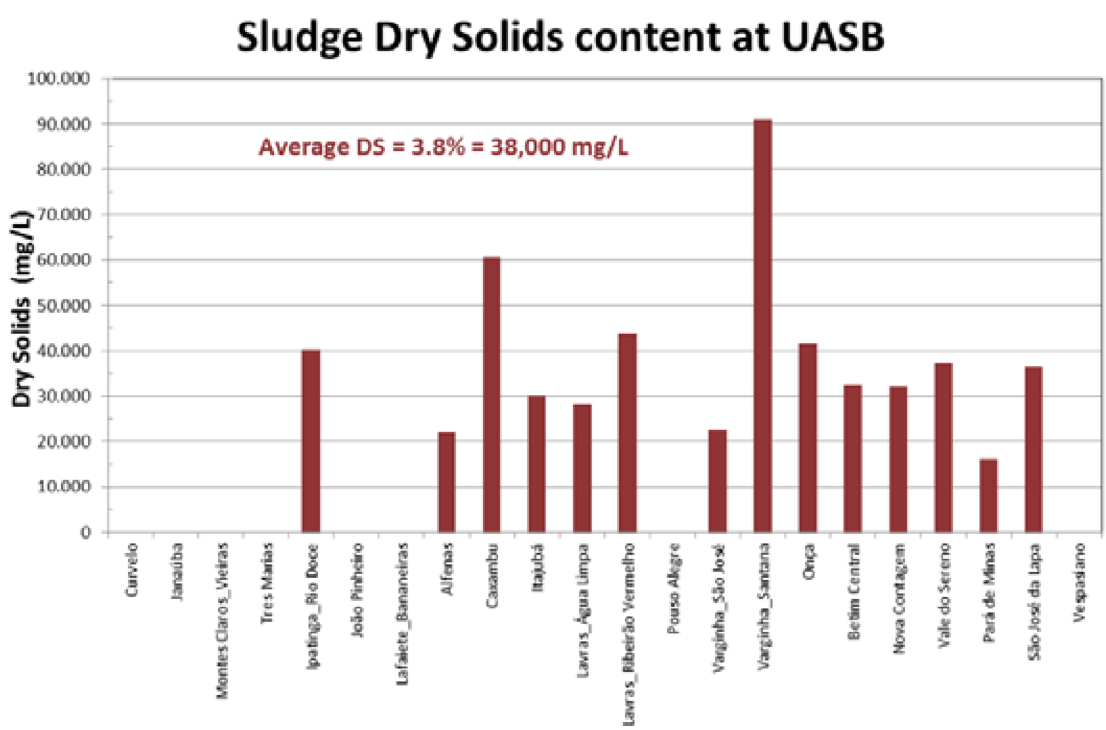
			Actual	Design
General				
Daily flow rate	average	m3/d	12,093	21,519
BOD_5 load	average	kg/d	3,745	6,981
COD load	average	kg/d	8,294	—
TSS load	average	kg/d	3,493	—
BOD_5 effluent	average	mg/L	62	53
COD effluent	average	mg/L	194	—
TSS effluent	average	mg/L	66	—
WASTEWATER TRAIN				
UASB retention time	average	h	15.8	7.8
UASB volumetric load	average	$gBOD_5$/m3/d	580	1,080
UASB surface load	average	m/h	0.4	0.6
SLUDGE TRAIN				
Sludge from UASB	average	m3/d	n.a.	n.a.
		kgDS/d	n.a.	n.a.
VS of raw sludge	average	%	n.a.	n.a.
Digester retention time	average	d	n.a.	n.a.
VS destruction in digester	average	%	n.a.	n.a.
VS of UASB sludge	average	%	56	n.a.
BIOGAS TRAIN				
Biogas production	average	m3/d	n.a.	n.a.
Specific biogas product.	average	L / $kgVS_{destroyed}$	n.a.	n.a.
Calorific value biogas	average	kWh/m3	n.a.	n.a.
H_2S	average	ppm	n.a.	n.a.
Wastewater temperature	average	°C	23.6	n.a.

Source: FWT 2013c.
n.a. = not available.

Sludge data quality at the investigated UASB systems was generally not perfect, and not all values of interest were available at all facilities. Figures CS3-4 and CS3-5 summarize important results as they were available.

Total sludge quantity equaled about 30 $gDS/PE_{60}/d$. These results, though, are just based upon a few values, since sludge quantities are poorly documented.

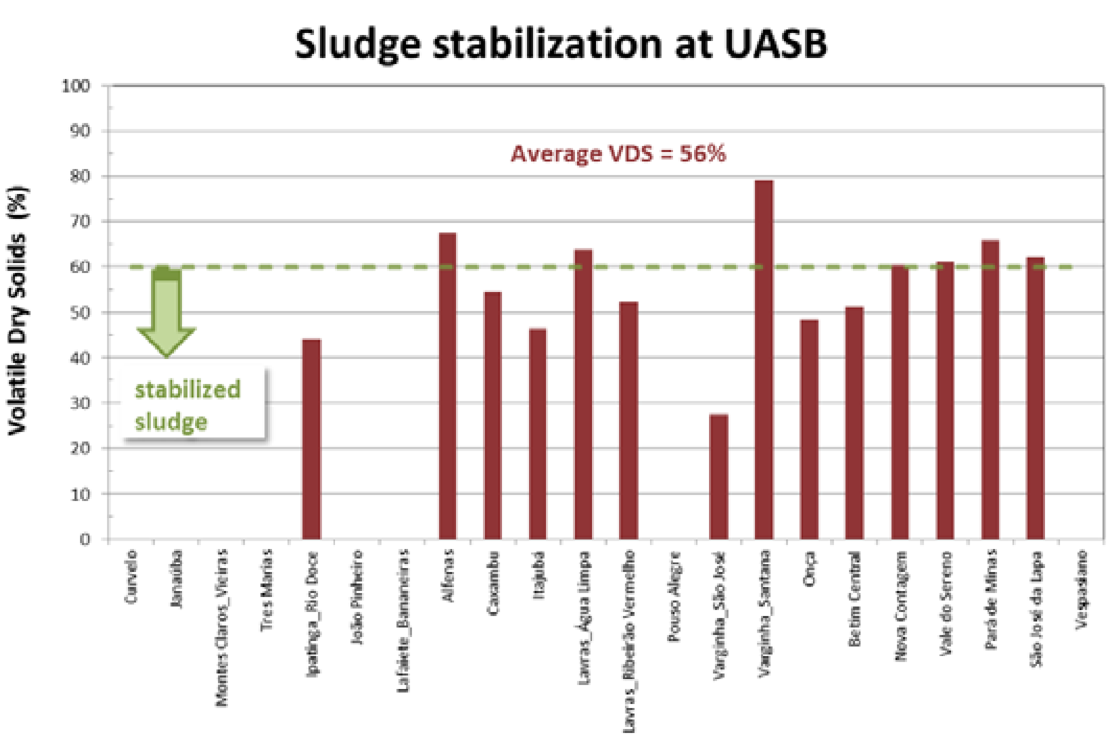
Figure CS3-4: Dry solids content of UASB sludge at WWTPs of case study 3



Source: FWT 2013c.

Note: Where no dry solids operation data are presented, none were available.

Figure CS3-5: Volatile solids content of UASB sludge at WWTPs of case study 3



Source: FWT 2013c.

3.2.2. Biogas production and potential for energy generation

Biogas production

Biogas flow metering was incomplete at the plants studied. The most reliable biogas quantification was done in the same project area by the local university in Minas Gerais (Universidade Federal de Minas Gerai [UFMG]) by Lobato and others (2011, 2012). The results of that assessment have been summarized in table CS3-11. Its plausibility has also been confirmed by results from other authors—for example, Noyola and others (2006)—whose results are also provided in that table. Here in this section, only the final conclusion is presented: mean collectable biogas is 13 L biogas/PE_{60}/d (range 7–17 L biogas/PE_{60}/d). This could be considered a good result, which is not much below what can be collected in an anaerobic heated state-of-the-art sludge digester. It is important to keep in mind, however, that this result is only possible in a properly designed and operated UASB.

Potential for energy generation

To date, none of the analyzed WWTPs has been producing electricity from biogas. Traditionally, the concept of energy generation from biogas had never been applied to UASB in the region. Moreover, it used to be impossible to supply electricity from a WWTP into the public grid. Most UASBs are not big energy consumers; hence, there was little incentive to invest in biogas utilization. The situation changed completely with a new federal law in 2012 allowing WWTPs to consume at one site the same electricity they supply to the grid at another (see section 3.2.4) at no cost. As a consequence, COPASA has now begun the process of assessing the biogas's energy potential and the financial implications of biogas utilization. It is to expect that several energy utilization projects will be implemented in the years to come.

Since no explicit, clearly defined biogas potential is available from the operators of the twenty-two UASBs, the subsequent analysis in case study 3 looks at a potential biogas range between 7 and 17 L/PE_{60}/d, with 13 L/PE_{60}/d considered the most likely practical result. This range of typically collectable biogas resulted from the analysis of different influent scenarios, as presented in table CS3-11. It can thus be seen as a good reflection of conditions prevailing for any of the twenty-two WWTPs of this case study. For the calculation of the electricity generation potential, the following values were additionally used: calorific value of the biogas = 6.5 kWh/m^3; electric efficiency of CHP = 30 percent. Both these assumptions are cautious, and an even higher electricity generation potential might be quite feasible in cases of higher calorific value and/or higher CHP efficiency. The results are summarized in table CS3-4.

Table CS3-4: Electricity production potential for a typical range of specific biogas production of 7–17 L/PE_{60}/d at UASBs

Electricity potential from biogas (kWh/PE_{60}/d)			
7 L/PE_{60}/d	10 L/PE_{60}/d	13 L/PE_{60}/d	17 L/PE_{60}/d
5.0	7.1	9.3	12.1

Source: Authors' calculation.

Note: L/PE_{60}/d x 16.67 = L/kg BOD_5/d.

A comprehensive database was available on the electric power requirements of all the analyzed UASB systems in the year 2011. The in-depth assessment showed a pronounced effect of economies of scale—that is, larger WWTPs consumed less energy, and vice versa. For practical presentation of the results, the plants were divided into four "classes" of different sizes, and for each class it was possible to derive median and benchmark energy consumption values (FWT 2013c); see table CS3-5 and figure CS3-6. The classes were defined as follows:

- Class 1: <10,000 PE_{60}
- Class 2: 10,000–50,000 PE_{60}
- Class 3: 50,000–100,000 PE_{60}
- Class 4: >100,000 PE_{60}

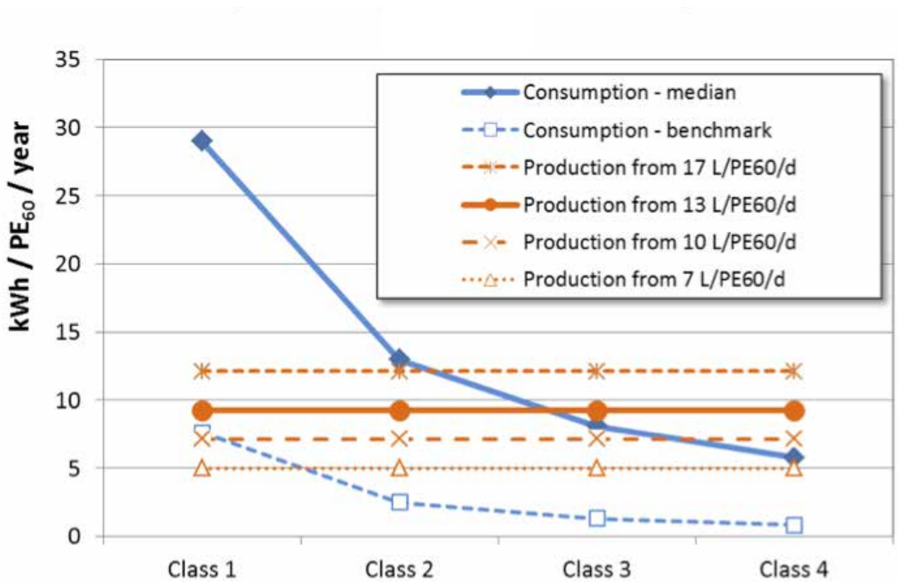
Table CS3-5: Electricity requirements of UASBs of case study 3

	PE_{60}	Electricity consumption median ($kWh/PE_{60}/y$)	Electricity consumption benchmark ($kWh/PE_{60}/y$)
Class 1	<10,000	29	7.6
Class 2	10,000–50,000	13	2.5
Class 3	50,000–100,000	8	1.3
Class 4	>100,000	6	0.8

Source: FWT 2013c.
Note: $kWh/PE_{60}/y \times 16.67 = kWh/kg\ BOD_5/y$.

The respective electricity requirements of the analyzed UASB plant classes are compared to the electricity production potential from biogas in figure CS3-6.

Figure CS3-6: Power generation potential of realistic biogas production range, compared to actual electricity consumption of analyzed UASB plants



Source: FWT 2013c.
Note: $kWh/PE_{60}/y \times 16.67 = kWh/kg\ BOD_5/y$.

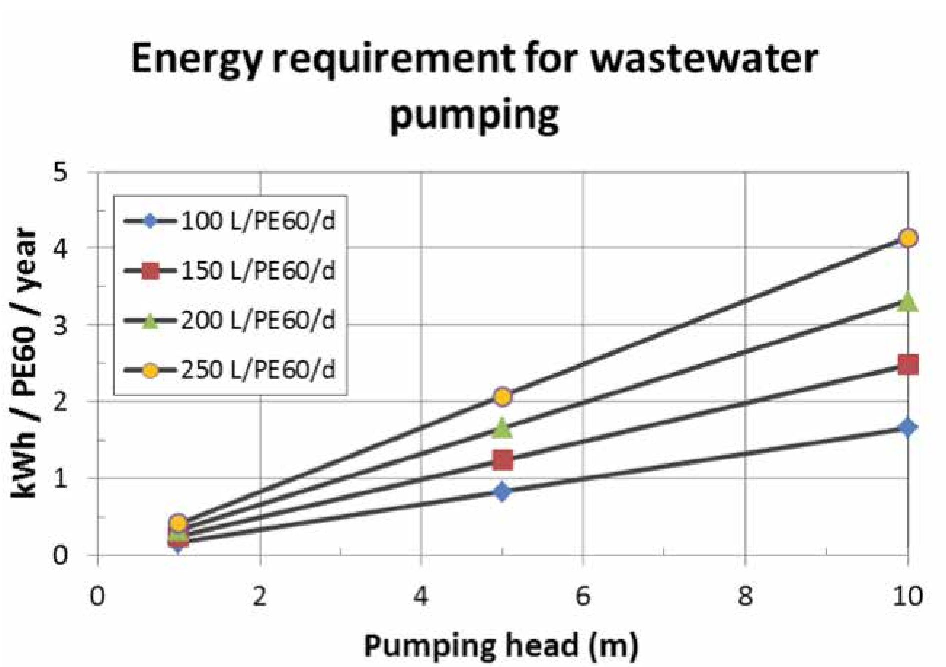
From these results, the following conclusions can be derived:

- Actual electricity consumption at UASB plants shows a clear effect of economies of scale: larger plants consume less energy, and vice versa.
- Median electricity consumption of large plants equals 6 kWh/PE_{60}/y; median plant sizes consume around 10 kWh/PE_{60}/y; and small ones consume a median of almost 30 kWh/PE_{60}/y.
- Benchmark values for the best performers have also been derived from the data. They are within 1–8 kWh/PE_{60}/y for all classes.
- No apparent correlation between specific electricity consumption and type of post-treatment could be found, apart from the observation that the only CAS polishing system had higher than average energy requirements. Hence, the electricity consumption at a specific plant appears to be dominated by the extent of wastewater pumping and by the electric efficiencies of the particular plant's installations. There was only one notable exception: the UASB system that used a CAS system for polishing featured elevated electricity consumption. However, in that specific case, this was not so much dependent on technology as it was a consequence of too-large blowers running at much overcapacity, thereby needlessly consuming too much energy.

Figure CS3-7 clarifies further by looking at energy requirements for wastewater pumping in general.

As it turns out, the energy requirement for pumping wastewater up to five meters is typically <2 kWh/PE_{60}/y, depending on the PE_{60} specific wastewater flow rate.

Figure CS3-7: Electricity requirement for wastewater pumping



Source: Authors' calculation.

Note: kWh/PE_{60}/y x 16.67 = kWh/kg BOD_5/y.

The energy consumption values presented in figure CS3-6 are reasonable, and benchmark values of 1–8 kWh/PE_{60}/y are feasible. To date, no comprehensive energy data on UASB, comparable to the content of figure CS3-6, have been published. These data are assumed to be the first benchmarking exercise for UASB. Electricity consumption data of UASB plants are usually not well documented nor published systematically.

When comparing the above electricity consumption values for large plants to that of CAS (case study 1), which equaled about 30–35 kWh/PE_{60}/year, and that of a large TF plant (case study 2), which equaled 9–10 kWh/PE_{60}/year, UASB plants apparently show similar energy needs to TF plants, with a tendency to even lower electric power consumption.

Generally it appears that, for medium plant sizes of about 50,000 PE_{60} upwards, UASB plants could operate with a positive energy balance on an annual basis if biogas production, collection, and utilization worked normally—that is, these plants will be able to produce more energy than they consume. The electricity supply from the public grid hence only serves as "safety net" for peak periods and during operational problems. Small UASB plants of less than 10,000 PE_{60}, on the other hand, will only achieve a positive energy balance if they are optimized both in terms of energy consumption and biogas production/utilization. Yet, in reality, it is more likely that these plants will only recover about 30–50 percent of their electricity requirements from biogas-generated sources. The biogas and power generation potential of UASB is summarized in table CS3-6. Note that in this case, the plants were all designed for carbon removal only.

Table CS3-6: Biogas and power generation potential of UASB plants of case study 3

	Key energy values of the investigated UASBs
Biogas production	
- N elimination (L/PE_{60}/d)	**—**
- C elimination (L/PE_{60}/d)	**avg. 13 (7–17)**
Electric efficiency CHP (%)	30
Thermal efficiency CHP (%)	55
Calorific value of biogas (kWh/m^3)	6.5
Power generation	
- N elimination (kWh/PE_{60}/year)	**—**
- C elimination (kWh/PE_{60}/year)	**avg. 9 (5–12)**
Thermal energy generation	
- N elimination (kWh/PE_{60}/year)	—
- C elimination (kWh/PE_{60}/year)	avg. 17 (9–22)
Total energy generation	
- N elimination (kWh/PE_{60}/year)	—
- C elimination (kWh/PE_{60}/year)	avg. 26 (14–34)

Source: Authors' calculation.

Note L/PE_{60}/d x 16.67 = L/kg BOD_5/d; kWh/PE_{60}/y x 16.67 = kWh/kg BOD_5/y.

3.2.3. Operation capacity needs, biogas safety

The key operational issues concerning UASB reactors relate to specific operational problems and to safety concerns due to the management of explosive biogas. The following are the most relevant aspects:

- Safety concerns
- Deposits in the digester
- Insufficient biogas treatment
- Scum formation

Figure CS3-8: Sand deposits inside UASB reactor

Source: Morais et al. 2013.

The first three issues have already been discussed in case study 1. The arguments and information provided there also prevail for case study 3. For details, see section 1.2.3.

Deposits in the digester

As indicated, the same problems and counter-remedies prevail in principle for UASB as for mesophilic sludge digesters (described in section 1.2.3). For instance, Morais and others (2013) describe sand accumulation of 1.5 meters depth inside UASB reactors due to malfunctioning of the grit chamber (see figure CS3-8).

Scum formation

This is a rather specific phenomenon observed in many UASB reactors worldwide. Only with a renewed interest in optimized biogas collection and utilization, though, is it nowadays receiving due attention. For many years it was just considered an operational hassle that was rarely investigated further. However, now it has become standard knowledge that substantial scum formation under the three-phase separator hampers biogas collection and can eventually even bring it to a standstill.

Gas bubbles apparently have difficulty penetrating the rather compact scum layer. The gas accumulates underneath and then eventually manages to escape into the open air via the sedimentation compartment or through small cracks and/or openings in the three-phase separator. Consequently, it is not surprising that the flares at many UASBs went out of operation long ago after the initial operation period, since no gas arrived there anymore.

The scum is a mixture of sludge, solids, and FOG (fat, oil, and grease; Chernicharo et al. 2013). Its accumulation rate is relatively high. Morais and others (2013) report a scum yield of 0.04 $L/kgCOD_{applied}$. Chernicharo and others (2013) report scum yields of 0.20–0.24 $L/m^2/d$, equivalent to 0.004 $L/kgCOD_{applied}$. This results in a scum buildup of about 1–10 mm per week under the three-phase separator. Hence, without scum removal, the scum layer will be several centimeters thick within a few weeks at the latest and will continue growing and compacting. Over time, manual removal becomes increasingly difficult, and removal by suction trucks proves time consuming and costly. Two or three workers can be fully employed all year round at a medium or large UASB plant just removing the scum. Some impressions regarding scum and its removal are presented in figure CS3-9.

If the scum is not removed from under the three-phase separator for prolonged periods, eventually it will also enter the sedimentation compartment and deteriorate even the effluent quality.

Figure CS3-9: Scum formation in UASB reactors

Scum removal at UASB Pará de Minas, Brazil

Scum removal at UASB Onça, Brazil

Scum in effluent compartment at UASB Nathay, Egypt

Scum under three-phase separator at UASB Onça, Brazil

Scum under three-phase separator at UASB Ipatinga, Brazil

Any counter-measure against scum can be based on one or both of the following elements:

- Prevention/minimization of scum formation
- Regular removal of scum

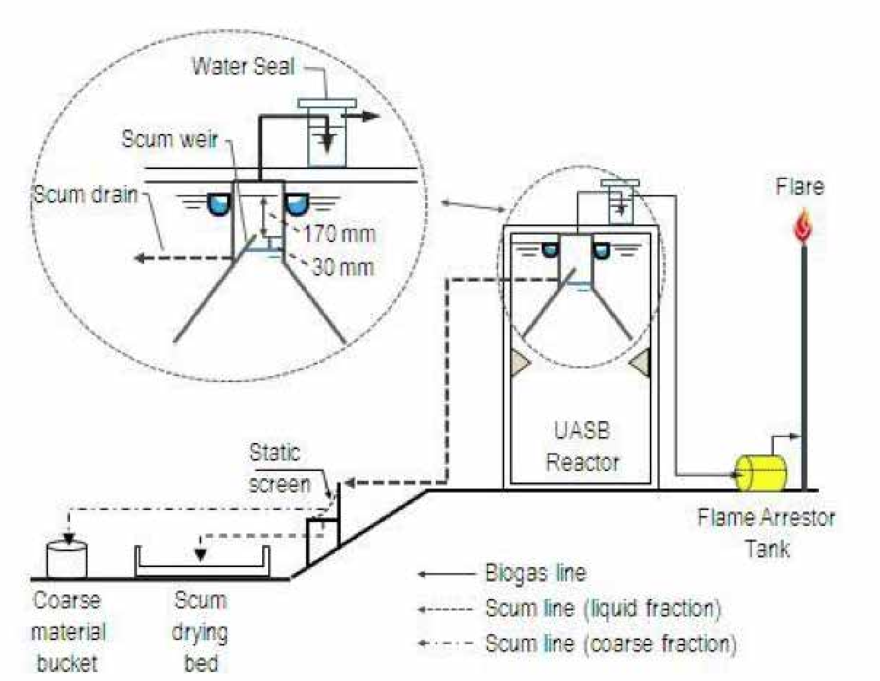
The *prevention/minimization of scum* must target its main constituents: (a) solids and (b) FOG.

For improved solids removal, apparently even 6 mm fine screens are insufficient. Many of the WWTPs investigated in case study 3 have screens, but they still suffer from scum. Therefore, it is recommended to use even finer screens. Ideally suited would be sieves with openings of 1 mm or 2 mm. The 1.0 and 1.5 mm sieves, for instance, have proved very effective in the elimination of almost all solids at MBR plants.

FOG removal is more difficult but nonetheless important. Parravicini (2012) describes the impact of FOG: these compounds attach to the sludge pellets and thus reduce their specific weight. Consequently, the sludge particles float easily and are incorporated into the scum on the liquor surface. None of the plants in case study 3 features a specific fat removal stage prior to the UASB stage. This also holds true for most other municipal UASB plants worldwide. There are conventional technologies to remove fat—for instance, through an aerated grit chamber. However, the efficiency of these installations is considered unreliable under warm climate conditions. After all, temperature is an ideal "solvent" for FOG. The authors of the present study believe a fine sieve, as described above, can have a similar effect: it is known that removing most of the solids also removes considerable quantities of the FOG attached to those solids. Thus, additional fat removal stages are definitely recommendable but might not be imperative if fine sieves are installed.

Scum removal must be automated if it is to work satisfactorily. Only if scum is removed regularly at brief intervals, when it is still quite liquid, can the operator avoid its solidification and compaction. One system suggested by Chernicharo and others (2013) is depicted in figure CS3-10: a weir is installed in the upper part of the three-phase separator, and the water level is kept 30 mm below the weir's crest. Opening a valve connected to this zone causes the pressure under the three-phase separator to drop and the water level to increase. Scum flows over the weir and is removed. The scum is sieved, with the liquor phase going to a drying bed and the solids going to a sanitary landfill. The authors report that the minimum scum removal efficiency was 80 percent, while over 90 percent was most common.

Figure CS3-10: Scum removal technology for UASB



Source: Suggested by Chernicharo et al. 2013.

3.2.4. Institutional aspects, energy costs

The institutional background for biogas utilization underwent a major change recently in Brazil. Before, it was legally impossible to supply electricity from a WWTP into the public grid; consequently, interest in biogas production and utilization was low, and most biogas from UASBs was just flared. In 2012, the situation changed completely: ANEEL, the national energy agency in Brazil, introduced legislation that allows the supply of electric power to the public grid from microgeneration sources (ANEEL 2012). Furthermore, a supplier to the public grid is now also allowed to withdraw the same quantities of electric power at another site, free of supply cost.

This means an operator of a WWTP can now substitute the unit cost of its own electricity consumption through autogenerated electricity. If one and the same operator is in charge of several plants, which is frequently the case in Brazil, it can supply the excess power from one WWTP into the public grid and withdraw it at another that does not dispose of electricity generation from biogas. Hence, it has become relatively easy for a WWTP operator to utilize 100 percent of the electric power generated from biogas at its own installations.

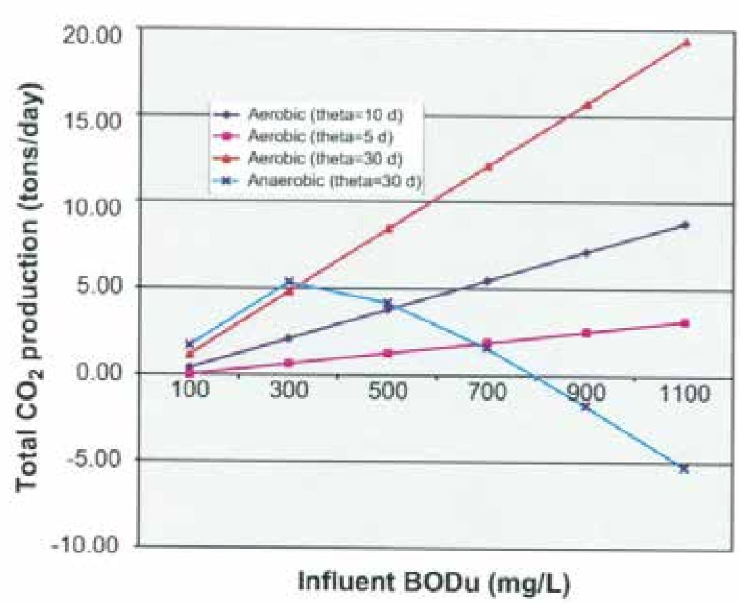
The financial OPEX gain of a WWTP operator producing its own electric power is thus equal to the cost of electricity purchased from the public grid. This unit cost per kWh is not exactly the same for any plant in Brazil, but the average of the twenty-two WWTPs investigated for case study 3 in 2011 was found to be US$0.25/kWh.

3.2.5. GHG reduction and CDM co-financing

For general information on CDM co-financing, price of carbon credits, and so on, see section 1.2.5.

An interesting question when summing up all GHG emissions of wastewater treatment is whether anaerobic treatment emits more or less GHG than aerobic treatment. Cakir and Stenstrom (2005) analyzed this issue and came up with an interesting result. The thinner influent wastewater is, the more CH_4 is lost in the treated effluent of anaerobic wastewater treatment. And since CH_4 is a twenty-one times stronger GHG than CO_2, this results in high CO_{2e} emissions from anaerobic wastewater systems. Consequently, anaerobic wastewater treatment has turned out frequently to emit more GHG than aerobic technologies, even though its energy consumption is low. The break-even point was calculated at an influent BOD_u of 300 mg/L for extended aeration and an influent BOD_u of 500–700 mg/L for aerobic wastewater treatment with sludge age between five and ten days. Only if influent BOD_u is higher than those values does anaerobic treatment lead to lower GHG emissions than aerobic treatment. Since municipal wastewaters frequently are not that concentrated, overall anaerobic systems emit more GHG than aerobic technologies (see figure CS3-11). This picture would change, though, if the CH_4 contained in the effluent of anaerobic systems were recovered.

Figure CS3-11: Total CO_{2e} emissions of wastewater treatment in function of influent BOD_u



Source: Cakir and Stenstrom 2005.

Note: Theta = sludge retention time; BOD_u = ultimate BOD.

Specifics of case study 3

As pointed out in section 1.2.5, due to the "additionality" criterion, the new construction of a UASB complete with CHP could only take into consideration the energy production component under CDM but not methane elimination, since the methane is created and eliminated by the project.

Brazil's specific energy mixture causes the GHG emissions from electricity generation presented in table CS3-7.

Table CS3-7: GHG emissions per kWh for electricity generation in Brazil, as compared to the world

Region	1990 gCO_2/kWh	2010 gCO_2/kWh
Brazil	55	87
World	586	565

Source: IEA 2012.

Note: The table shows CO_2 emissions from fossil fuels consumed for electricity generation, in both electricity-only and combined heat and power plants, divided by output of electricity generated from fossil fuels, nuclear, hydro (excluding pumped storage), geothermal, solar, wind, tide, wave, ocean, and biofuels. Both main activity producers and autoproducers have been included in the calculation.

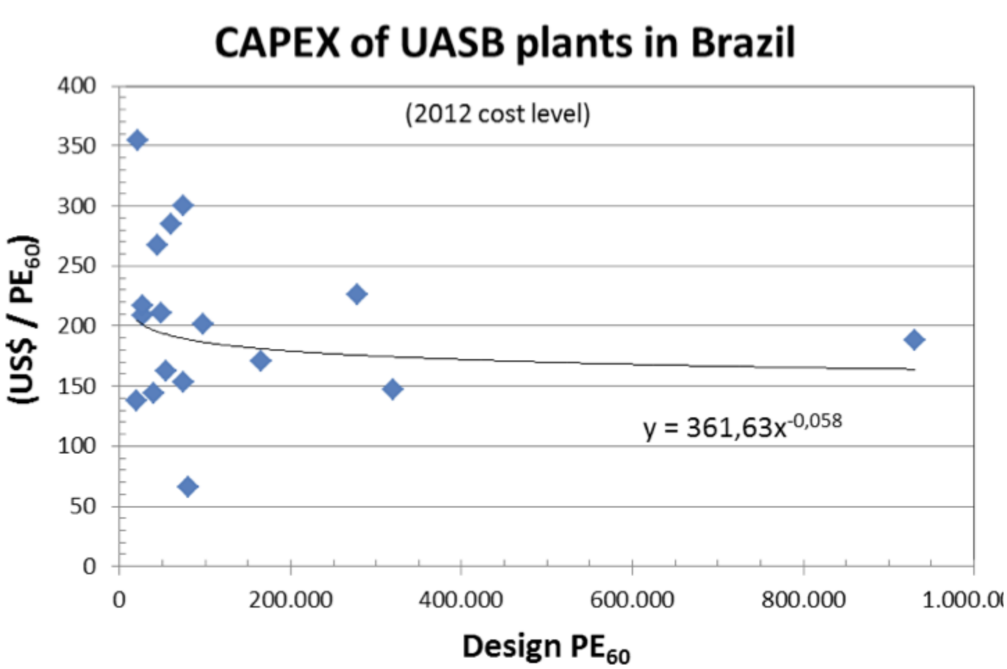
The annual electricity production potential from biogas of the twenty-two WWTPs of case study 3 equals 12.7 million kWh/year. This would allow a potential GHG reduction of 1,100 tons CO_{2e}/year for case study 3 in 2011, based on Brazil's average energy mixture.

3.2.6. CAPEX structure

Complete UASB plant

CAPEX of seventeen of the twenty-two WWTPs of case study 3 was available as total CAPEX for each and every plant. COPASA updated these CAPEX to 2012 cost levels, which were then related to the individual design capacity of each plant (FWT 2013a). This resulted in the cost curve presented in figure CS3-12.

Figure CS3-12: CAPEX of UASB plants of case study 3



Source: FWT 2013a.

Note: US$/$PE_{60}$ x 16,67 = EUR/kg BOD_5.

Comparing these values to international CAPEX information leads to different results. For instance, Libhaber and Orozco-Jaramillo (2012) mention CAPEX of US$20–40/capita for UASB reactors. Wagner (2010) reports about US$25–50/capita for UASB + ponds and specific UASB reactor cost of US$150–250/$m^3$. WERF (2010b) cites CAPEX for UASB + post-treatment of US$35–60/PE. Sperling and Chernicharo (2005) indicate a range of about US$15–45/capita for UASB + post-treatment.

All in all, the literature points toward CAPEX of <US$60/capita. Assuming a typical raw wastewater loadings of 40 g BOD_5/cap/d in those countries to which the data refer, this CAPEX is less thanUS$90/$PE_{60}$, whereas the results from case study 3 deliver values mostly in the range of US$150–200/$PE_{60}$ (EUR110–150/PE_{60}).

Additional CAPEX for biogas utilization

The CAPEX required for installing a biogas utilization system has been analyzed for five of COPASA's UASB systems (ARAconsult 2013). The following outcomes were found:

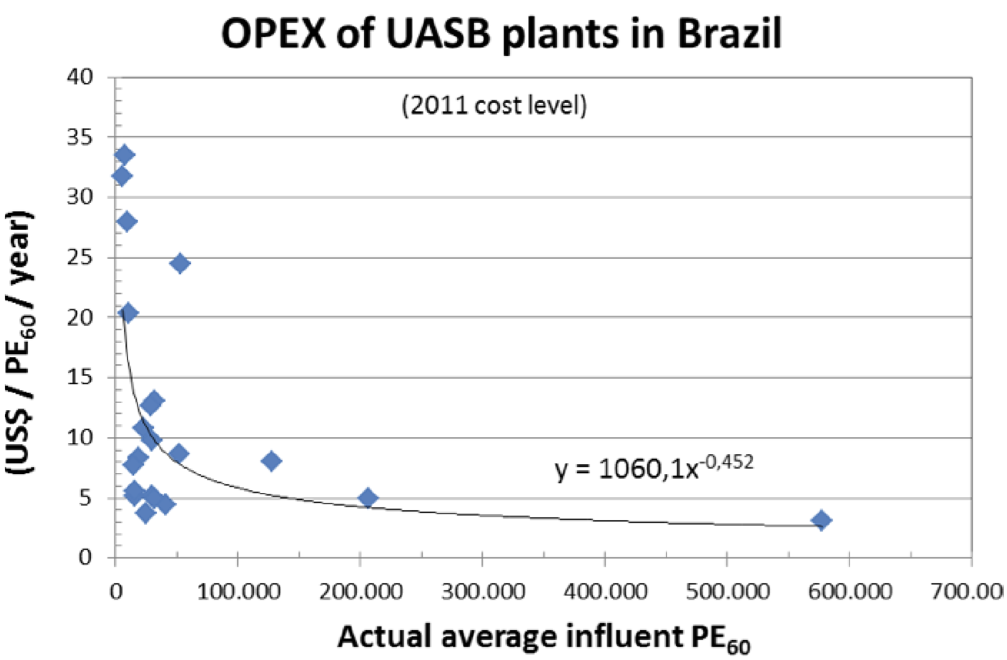
- New preliminary treatment, complete with fine sieves 1–2 mm and new aerated grit chambers for sand and FOG removal: CAPEX = avg. US$7.5 (range US$4–11)/PE_{60} = avg. EUR5.5 (EUR3–9)/PE_{60}.
- Biogas system: collection, treatment, storage, and utilization in CHP: CAPEX = avg. US$8.0 (US$5–15)/PE_{60} = avg. EUR 6.0 (EUR 4–11)/PE_{60}.
- Generally, there was a clear effect of economies of scale: smaller plants were more expensive, per population treated, than larger ones. Additionally, whether more or less CAPEX was required depended somewhat on the physical condition of the existing installations.

3.2.7. OPEX structure

Complete UASB plant

OPEX of all twenty-two WWTPs of case study 3 was available as total OPEX for each and every plant for the year 2011. These were then related to the individual average influent load of each plant (FWT 2013a), resulting in the cost curve depicted in figure CS3-13.

Figure CS3-13: OPEX of UASB plants under case study 3



Source: FWT 2013a.

Note: US$/PE$_{60}$/y x 16,67 = US$/kg BOD$_5$/y.

Comparing these values to international OPEX information leads to different results: Libhaber and Orozco-Jaramillo (2012) mention OPEX of US$1.0–1.5/capita/year for UASB reactors; Wagner (2010) reports US$1.2/capita/year for UASB + ponds in Bolivia; and Sperling and Chernicharo (2005) indicate a range of about US$1.5–5.0/capita/year for UASB + post-treatment.

The literature points toward OPEX of <US$5/PE$_{60}$/year, whereas the analysis from case study 3 delivers values <US$5/PE$_{60}$/year only for large WWTPs (<EUR3.7/PE$_{60}$/year), while medium plants <50,000 PE$_{60}$ mostly operate within the range of US$3–35/PE$_{60}$/year (EUR2–25/PE$_{60}$/year). The conclusion is that the cost level in Brazil is apparently higher than in many other locations. But it is also possible that the numbers in the literature do not properly reflect total OPEX in general, and/or are not up to date, and/or just relate to very large plants.

The financial value of the electric power potential can be assessed by applying the average electricity tariff for the twenty-two WWTPs investigated for case study 3 in 2011. This value is US$0.25/kWh.

Additional OPEX increments result from the new investment in the biogas system and its subsequent O&M requirements. (No additional OPEX is assumed for the change in preliminary treatment, where an existing system is merely replaced by a similar, though technically more efficient, system.)

The overall OPEX change due to biogas utilization is summarized in table CS3-8.

Table CS3-8: Average OPEX change due to biogas utilization projects at the 22 WWTPs of case study 3

	Analysis	OPEX 2011	
		US$/y	EUR/y
Biogas system: additional OPEX for O&M	2 percent of 8.00 US$/PE$_{60}$*	409,573	303,387
Saving due to electric power production	-12,700,000 kWh/y with 0.25 US$/kWh	-3,175,000	-2,351,852
TOTAL		**-2,765,427**	**-2,048,465**

Source: Authors' calculation.
Consequently, OPEX will reduce by US$2.77 million/y (EUR2.05 million/y).

3.2.8. Viability of investment in biogas utilization

Table CS3-9 summarizes several cost indicators for case study 3.

Table CS3-9: Cost indicators for biogas utilization project at all 22 UASBs of case study 3

		2011
Average influent load	$PE_{60,avg}$	1,373,166
Peak influent load	$PE_{60,max}$	2,559,829
CAPEX	US$	20,478,633
OPEX	US$/y	-2,765,427
specific CAPEX	US$/$PE_{60,avg}$	14.91
	US$/$PE_{60,max}$	8.00
specific OPEX	US$/$PE_{60,avg}$/y	-2.01
	US$/$PE_{60,max}$/y	-1.08

Source: Authors' calculation.
Note: 1 PE_{60} = 1.11 PE_{54} = 1.11 cap in Brazil; US$/$PE_{60}$ x 16.67 = US$/kg BOD_5.

The specific CAPEX for biogas utilization projects at all twenty-two UASBs of case study 3, related to the plants' design capacity, is US$8.0/$PE_{60,max}$ (EUR6/$PE_{60,max}$). The specific CAPEX related to the average actual influent pollution is US$14.9/$PE_{60,avg}$ (EUR11.0/$PE_{60,avg}$).

Overall, the biogas project will reduce OPEX. This savings equals US$1.1/$PE_{60,max}$ (EUR0.8/$PE_{60,max}$), or US$2.0/$PE_{60,avg}$ (EUR1.5/$PE_{60,avg}$).

Therefore, the investment in new biogas utilization systems for all of this case study's UASBs will be paid back within about seven years (without interest rates). This can be considered financially viable. The lifespan of the new investment is usually assumed to be about twelve to fifteen years, which is almost double this period.

3.3. Conclusions for UASB in EAP countries

What remains unchanged in EAP (as compared to case study 3)

The key characteristics of UASB described in case study 3, such as BOD_5 elimination, VS destruction, sludge characteristics, and biogas production, can be transferred from Brazil to EAP countries without major changes, since climate conditions are similar.

What changes in EAP (as compared to case study 3)

One change possibly needed in EAP relates to wastewater treatment requirements. The UASB systems in Brazil are designed for carbon removal only. If, in EAP, additional nitrification is required, more efficient polishing stages will be needed. This will generally push up CAPEX, and it will increase OPEX, particularly where CAS is applied for nitrification. For polishing trickling filters (TFs), OPEX will stay the same. After all, the pumping head does not change when more TFs need to be supplied to facilitate nitrification.

If denitrification is required as well, the additional installations will, in any case, increase CAPEX, OPEX, and energy consumption. This might eventually lead to a situation where, with nitrification plus denitrification, less than 100 percent of electricity consumption can be covered from biogas, even at large UASBs. Cao (2011) reported on pilot experiments with UASB + CAS in Singapore. His findings allow the conclusion that electricity coverage of up to about 70 percent would be feasible with nitrification and denitrification. Cao also pointed to the general problem of lacking carbon sources for denitrification after UASB. This is not to say the UASB + denitrification would not work, but denitrification downstream of very efficient BOD removal is generally challenging.

It is not clear, though, if the same amounts of biogas can be collected in cases of diluted wastewater, as is common in EAP. Due to the higher flow rate per capita, a higher percentage of the generated methane can be lost—dissolved via the liquid effluents—even if the same quantities of biogas are generated. This will have a negative impact on the electricity generation potential, and it will increase the GHG emissions. Tentatively, it may be realistic to assume a collectable biogas potential and, thus, electricity generation potential at the lower end of the ranges indicated in case study 3, as shown in table CS3-6.

Dilution of wastewater also has negative impacts on UASB design. One of the main design parameters is retention time in the UASB reactors. Increasing flow rates thus lead to increasingly large (and expensive) UASBs.

It is also possible that the quantities of digested sludge will be lower in EAP than in Brazil. This is particularly expected where septic tanks continue to be used in large numbers. Septic tanks retain solids present in wastewater, thereby reducing the amount of solids entering a WWTP in the first place. Inevitably, this will lead in EAP to reduced quantities of digested UASB sludge.

CAPEX is expected to be mostly lower in EAP than in Brazil, due to higher prices in Brazil. But OPEX savings will also be lower in EAP, both because of the reduced power production potential and because power unit cost is mostly lower in EAP.

Success stories in Asia

In South Asia, India can be considered the leader in UASB technology, with several large UASB plants already in operation for many years. Apart from the better-known 5 MLD and 36 MLD UASB plants in Kanpur, which were among the earliest large-scale UASBs worldwide, UASBs were also constructed under the Ganga Action Plan in Jajmau and Mirzapur. Furthermore, under the Yamuna Action Plan, ten UASB plants were constructed in Haryana in the towns of Faridabad, Gurgaon, Karnal, Nagar, Panipat, and Sonipat, all with design capacities of 10–50 MLD. A 60 MLD UASB plant exists in Ujjain, constructed under the Kshipra River Pollution Abatement Scheme (www.iramconsult.com, World Bank 2010). Some of those systems included CHP, but the design was poor, and the technology of biogas treatment was expensive. Hence, as far as is known by the authors, most of these biogas systems were taken out of operation. Generally, in terms of wastewater treatment, these systems proved quite reliable, even when receiving problematic industrial wastewaters, but they suffered from bad material quality and, hence, corrosion, sometimes even aggravated by the characteristics of industrial discharges to the sewer system.

UASB: TECHNICAL SUMMARY

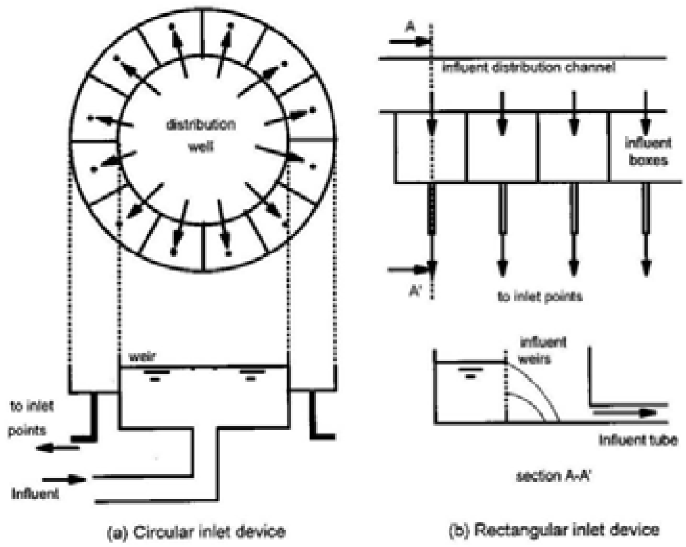
Table CS3-10: UASB: General design parameters and key characteristics

UASB DESIGN

- Retention time =
  - 6 hours for tropical countries (2–14 hours in general) (Haandel and Lettinga 1994)
  - 8 hours (WERF 2010b)
  - 10–14 hours (16–19°C), 6–9 hours (20–26°C), <6 hours (>26°C) (Sperling and Chernicharo 2005)
- Volumetric BOD load = <1,750–2,000 g $BOD_5/m^3/d$ (Haandel and Lettinga 1994; Sperling and Chernicharo 2005); 1,650 at 20°C and 2,700 at 25°C (Libhaber and Orozco-Jaramillo 2012)
- Upflow velocity:
  - <1.0 m/h (Haandel and Lettinga 1994; Libhaber and Orozco-Jaramillo 2012)
  - <0.6 m/h (WERF 2010b)
  - <0.5–0.7 m/h, max. 1.5–2.0 m/h (Sperling and Chernicharo 2005)
- Sludge production:
  - UASB sludge: 0.2–0.4 $gDS/gBOD_{5,\ influent}$ (Haandel and Lettinga 1994; Sperling and Chernicharo 2005)
  - Co-digested secondary sludge from polishing stage; see information provided in tables CS1-4 and CS1-5. An average sludge retention time of ffi30 days can be assumed in UASBs.
- Typical sludge DS = 3–5 percent (Sperling and Chernicharo 2005)
- BOD removal efficiency: Is calculated by equations, such as the following:
  - Haandel and Lettinga 1994: $E_{BOD} = 100 \times (1 - 0.68 \times t^{-0.68})$ for T > 20°C
  - Sperling and Chernicharo 2005: $E_{BOD} = 100 \times (1 - 0.70 \times t^{-0.50})$ for T = 20–27°C
  - The BOD removal efficiency at T < 20°C is still subject to discussion. For instance, Singh and Viraraghavan (2003) recommend T = 15°C to reduce the efficiency calculated for 20°C by 10 percent.
- SS removal efficiency: No reliable design tool exists, but an assumption of SS in the UASB effluent of 40–100 mg/L appears appropriate.

UASB SHAPE AND CONSTRUCTION	Both rectangular and circular UASB reactors have been constructed. However, the rectangular type is by far the most widespread shape nowadays. Typical water depth in UASB is in the range of 3–6 m. The total reactor volume is usually split over several reactors. The individual reactor volume per unit recommended was <1,000 m^3 (Haandel and Lettinga 1994) in the '90s; nowadays it is 2,000–3,000 m^3 per unit at large WWTPs. The environment in UASBs is highly corrosive. Consequently, most UASB reactors are constructed out of reinforced concrete, possibly with sufficient chemical resistance. It is imperative that all equipment that is not made from concrete, ranging from pipes, channels, distribution chambers, and manhole covers to the gas collection system, be made from noncorrosive materials such as PE, PVC, or GRP. The main construction differences of UASB found nowadays among large-scale plants relate to the influent distribution.
INFLUENT DISTRIBUTION	It is of major relevance that the influent wastewater be evenly distributed at the bottom of the UASB reactor. To that end, the total incoming flow is split into numerous subflows, each of which is then discharged at the UASB bottom through a specifically assigned pipe. The influence area of this supply pipe should be less than 5 m^2, and ideally between 1.5 and 3.0 m^2 (Sperling and Chernicharo 2005). With increasing flow rates, flow splitting is increasingly difficult to implement. Hence, influent distribution becomes a limiting factor in terms of the maximum treatment capacity UASB plants can achieve. For example, Onça WWTP (which is also part of this case study) is designed for a peak flow of 3.7 m^3/s (avg. 2.1 m^3/s), which is near the maximum flow range of what is practically recommendable. Wastewater splitting is usually done on the reactor surface, either by circular distribution devices or longitudinal (rectangular) devices (see figures CS3-14 and CS3-15). The former consist of circular distribution wells, and the latter use longitudinal free surface channels along which numerous distribution boxes are located. Each distribution box then receives its specific distribution pipe. To minimize clogging risks, the pipes should have a diameter of >75 mm. Since each pipe is hydraulically separated from the other, clogging can be easily spotted. In that case, pipe cleaning is done either with mechanical tools or high-pressure water.

Figure CS3-14: Typical distribution systems of influent to UASB



Source: Haandel and Lettinga 1994.

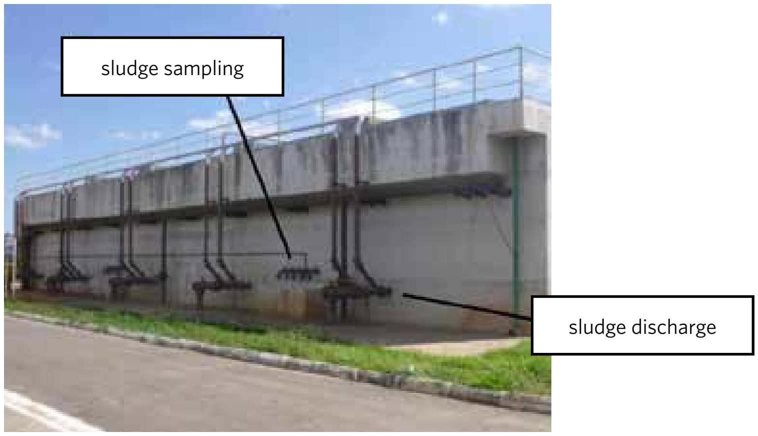
Figure CS3-15: Circular and rectangular inlet distribution at Betim Central WWTP (Brazil) and Cafeco WWTP (El Salvador)

Source: COPASA—Betim Central WWTP, CAFECO WWTP.

SLUDGE SAMPLING AND DISCHARGE

It is important for the operator to know the distribution of the sludge blanket inside the UASB and to optimize the discharge of sludge (as constant as with as high TS concentration as possible). To that end, it is advisable to install sludge sampling and discharge pipes at different levels of the reactor. Sampling is usually done at several levels, 0.5–1.0 meters apart, while sludge discharge is often possible at just two levels: one near the reactor bottom and another 1.0–1.5 meters above that level (see figure CS3-16).

Figure CS3-16: Sludge sampling and discharge pipes at Montes Claros WWTP (Brazil)



Source: COPASA—Montes Claros WWTP.

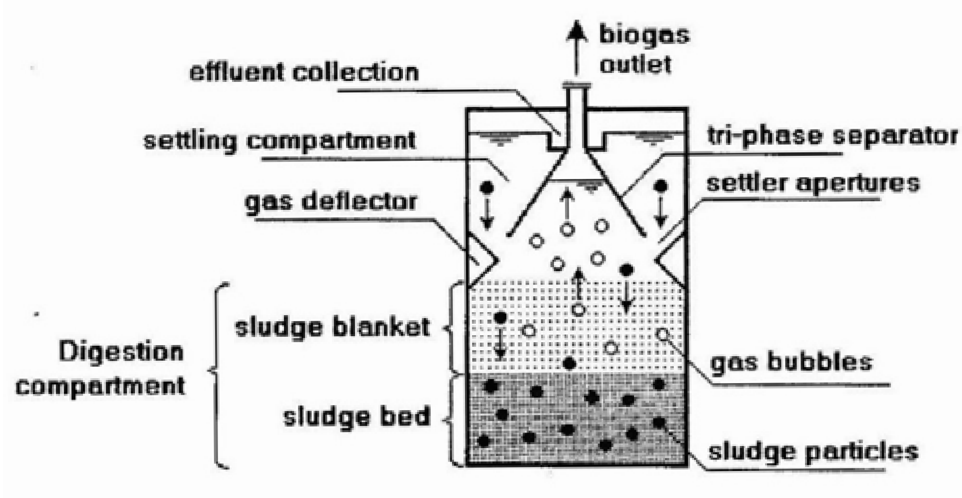
Three-phase separator The three-phase separator is a key component of any UASB and is located in the upper zone of the reactor. Its purpose is to separate the three phases: biogas, liquor, and sludge. After all, all three phases move upward, since this is the wastewater flow direction, and gas bubbles rise as well.

The three-phase separator (figure CS3-17) is constructed in such a way that the rising gas bubbles are collected and conveyed to the gas outlet pipe. This leads to a separation of gas and sludge particles, with the sludge particles consequently settling back into the sludge blanket. Still, some of the sludge particles may enter the settling compartment together with the treated effluent. Since the gas has already been effectively removed and cannot enter this compartment (as all vertical trajectories lead to the gas outlet), the sludge will be separated from the liquid phase here, and the particles will sink and glide back into the sludge blanket from this zone as well.

Design and construction details (Sperling and Chernicharo 2005; Libhaber and Orozco-Jaramillo 2012)

To facilitate the gliding back of solids from the sedimentation compartment to the digestion compartment, and for efficient gas collection underneath, the inclination of the three-phase separator should be at least 45°, and preferably greater than 50°. The sedimentation compartment itself should be 1.5–2.0 meters deep. Hydraulic retention time in the sedimentation compartment should be 1.5–2.0 hours on average (>1.0 [0.6] hours during daily [hourly] peak flows), and surface loading in the sedimentation compartment should be 0.6–0.8 m/h (<1.2 [1.6] m/h during daily [hourly] peak flows). The velocity in the apertures to the sedimentation compartment may not exceed 4.0 (5.5) m/h during daily (hourly) peak flows, and should be <2.0 m/h during average flow.

Figure CS3-17: Schematic section through an UASB reactor



Source: Sperling and Chernicharo 2005.

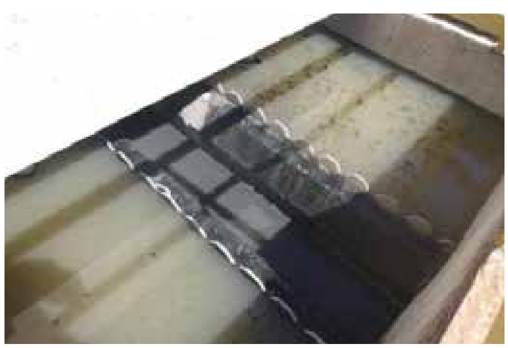
Three-phase separators can be constructed from quite different materials, ranging from coated metal sheets and plastic foils to concrete (see figure CS3-18).

Figure CS3-18: Three-phase separators

TREATED EFFLUENT COLLECTION

The treated effluent is collected on the reactor surface either through (a) launders with overflow V-notch weirs (figure CS3-19) or (b) submerged perforated tubes. The former have the advantage of easier cleaning and control but are rather sensitive to exact leveling. The latter are more difficult to clean, if needed.

Figure CS3-19: Treated effluent collection via launders with weirs at Pará de Minas WWTP (Brazil)



Source: COPASA— Pará de Minas WWTP.

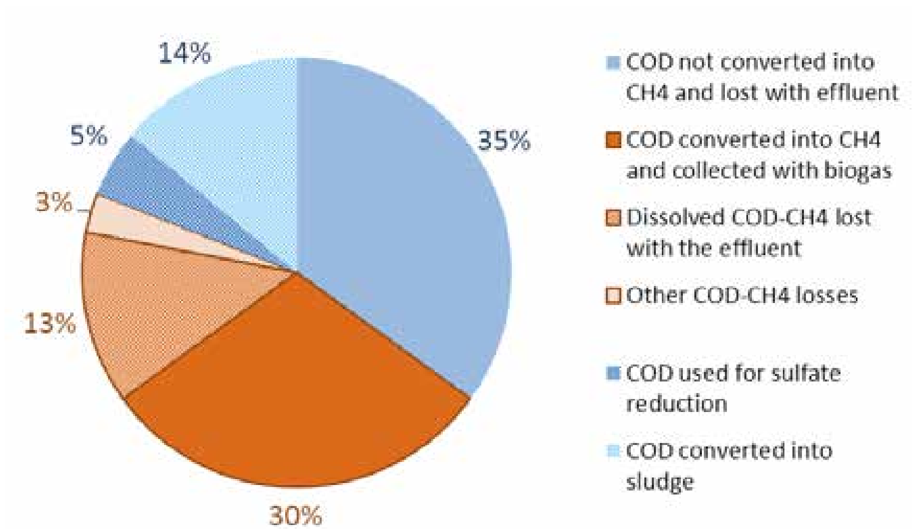
Table CS3-11: UASB biogas systems: General design parameters and key characteristics

BIOGAS DESIGN	• Volatile solids concentration of sludge: UASB sludge: avg. 55 percent (50–60 percent) (Bauerfeld et al. 2009) • Volatile solids destruction of sludge from polishing stages in UASB: according to principles presented in table CS1-5 • Gas production: - Traditionally, the biogas yield has been estimated in UASB through a simple COD balance: $COD_{biogas} = COD_{influent} - COD_{effluent} - COD_{sludge}$, combined with the general gas equation and an assumed CH_4 content in the biogas. This procedure is, for instance, described in Haandel and Lettinga (1994) and Sperling and Chernicharo (2005). However, in practice it turns out that this approach mostly leads to a more or less significant overestimation of the collectable biogas quantities. It is hence not recommendable. - Generally, it is of major importance to distinguish between biogas production and collectable biogas. There is always a considerable percentage of biogas that cannot be collected for various reasons. As has been found by Lobato and others (2011, 2012), the following requires careful analysis: COD converted into sludge; COD used for sulfate reduction; dissolved COD of CH_4 lost with effluent; COD not converted into CH_4 and lost with effluent; and other CH_4 losses. Based on an elaborate study of several UASBs, and also taking typical parameter variations into account, the following conclusion can be derived: collectable biogas = 10–17 L/cap/d (mean 14 L/cap/d). Behind these values are capita-specific pollution loads of an average 90–110 gCOD/cap/d. Hence, by including a small safety margin, it is possible to equate these capita-specific values with PE_{60}-specific values. On that basis, the following collectable biogas yield can be assumed: 10–17 L biogas/PE_{60}/d (mean 14). - It is important to mention that the result of the above-described latest findings related to collectable biogas yields match earlier findings. Behind the above-cited PE-specific gas production numbers of Lobato and others (2011, 2012) lies a CH_4 yield of 113–196 L CH_4/$kgCOD_{removed}$. This is rather similar to what had been found by Noyola and others (2006), who reported a range of 80–180 L CH_4/$kgCOD_{removed}$ (≈7–16 L biogas/PE_{60}/d).

- Conclusion: Based on the above described findings, the collectable biogas yield of municipal UASB can be expected to fall within a range of 7–17 L biogas/PE_{60}/d, with average values of 13 L biogas/PE_{60}/d (= 217 L biogas/$kgBOD_5$/d, range = 115–280 L biogas/$kgBOD_5$/d).

- Typical biogas characteristics: (a) methane CH_4: 60–80 percent; (b) calorific value: 6.0–8.0 kWh/Nm^3 (Sperling and Chernicharo 2005)
- COD mass balance: See typical example in figure CS3-20.

Figure CS3-20: Typical COD mass balance for UASB



Source: Lobato et al. 2011, 2012.

Typically, less than 50 percent of influent COD is converted into CH_4. The more than 50 percent of COD remaining is lost in the effluent, required for sulfate reduction, or converted into sludge. From the COD converted into CH_4 only about two-thirds are actually collectable in the UASB, while one-third is lost. These losses can even go up to about 50 percent of the CH_4 formed under unfavorable conditions. Under ideal conditions, they may be reduced to about 25 percent of the CH_4 formed (Lobato et al. 2011, 2012). Similar results are presented by Chernicharo and others (2012).

BIOGAS TREATMENT *See table CS1-5.*

GAS HOLDER *See table CS1-5.*

FLARE *See table CS1-5.*

BIOGAS UTILIZATION *See table CS1-5.*

CASE STUDY 4: COVERED ANAEROBIC PONDS

4.1. BACKGROUND, PROCESS DESCRIPTION

4.1. Background, process description

4.1.1. Data sources

Waste stabilization ponds (WSPs) are a simple and widespread wastewater treatment technology. Typically, they feature a series of ponds that cover considerable areas of land. Apart from pumping the wastewater into them, usually no electric energy is required for running those systems. For the EAP countries on which this report focuses, WSPs are well suited in terms of the warm climate. However, given that only municipal WWTPs are discussed here, the land requirement is a serious constraint, and, in most cases, classical WSP systems may not be viable as municipal wastewater treatment systems.

The first, high-loaded stage of WSPs—anaerobic ponds (APs)—could be a technology component to consider in terms of energy generation potential. The subsequent phases of facultative ponds and maturation ponds are less relevant from that point of view. This has held particularly true from the moment it was proved feasible to cover APs with plastic covers and collect the biogas generated underneath. The AP thus turns into a combined wastewater treatment and digestion unit. Its land requirements are relatively small as compared to complete WSP systems, and yet still more than 50 percent of organic load is removed. Subsequent treatment stages will be needed to bring effluent quality to required levels, but the effects of removing a considerable organic load free of energy cost, thereby producing and collecting biogas that can be used for electricity generation, is an attractive combination.

Case study 4 looks into the practical results from two cities in which covered anaerobic ponds have been applied for many years: Santa Cruz in Bolivia and Melbourne in Australia. All operation data utilized in this case study were provided by the plant operators: SAGUAPAC (Cooperativa de Servicios Públicos Santa Cruz Ltda), www.saguapac.com.bo, and Melbourne Water, www.melbournewater.com.au.

Santa Cruz is a fast-growing, economically dynamic city with a population of about 1.5 million, located in the tropical lowlands of Bolivia at about 17° south latitude and about 400 meters altitude. Its annual average air temperature is around 23°C.

Melbourne is the capital of the state of Victoria. With 4.3 million inhabitants, it is the second most populous city in Australia. Located at about 37° south latitude at sea level, it has an annual average air temperature of around 20°C.

4.1.2. Wastewater management

Santa Cruz, Bolivia

SAGUAPAC operates four WWTPs, based on WSP technology with biogas collection in APs. Since two plants are located together at the same site, there are just two sites for biogas utilization. The key features of those plants are summarized in table CS4-1, and some pictures are provided in figures CS4-1, CS4-2, and CS4-3.

Table CS4-1: Key characteristics of WWTPs in Santa Cruz, analyzed for case study 4

WWTP	Technology	Total ponds		Anaerobic ponds		
		number	area	number	water depth	area
		(nr.)	(ha)	(nr.)	(m)	(ha)
System North						
Norte 1	AP - FP - MP	5	20	2	3.5	3.0
Norte 2	AP - FP - MP	8	39	2	4.5	3.7
SUBTOTAL		**13**	**59.0**	**4**	**—**	**6.7**
System East						
Este	AP - FP - MP	12	50	3	4.5	3.8
Parque Indus-trial	AP - FP - MP	6	12	3	3.5	1.8
SUBTOTAL		**18**	**62.0**	**6.0**	**—**	**5.6**
TOTAL						
Average		**8**	**30**	**3**	**4.0**	**3.1**
Total		**31**	**121**	**10**	**—**	**12.3**

Source: SAGUAPAC 2014a.
Note: AP = anaerobic pond; FP = facultative pond; MP = maturation pond.

Both plants were equipped in 2009 with 2 mm rotary sieves for preliminary treatment. This was done to eliminate a maximum of coarse solid materials before the wastewater reached the APs (SAGUAPAC 2014a).

For details of actual wastewater characteristics and other parameters, see tables CS4-3 and CS4-4. For details regarding general design recommendations for APs, see table CS4-12.

Figure CS4-1: Covered APs with and without biogas at Santa Cruz WWTPs

Sources: www.saguapac.com.bo; SAGUAPAC 2008.

Figure CS4-2: System North of Santa Cruz WWTPs

Source: Google Earth.

Figure CS4-3: System East of Santa Cruz WWTPs

Source: Google Earth.

Melbourne, Australia

Melbourne has two WWTPs—"Western Treatment Plant" and "Eastern Treatment Plant"—which serve 1.6 and 1.5 million people, respectively. While Eastern Treatment Plant, started up in 1975, is a CAS system with a tertiary treatment stage, Western Treatment Plant has a much longer history and a unique treatment concept.

The history of Melbourne Water's Western Treatment Plant goes back to 1897, when an infiltration facility was put into operation. Later, in 1936, the first treatment lagoon was added to act as a polishing system for the infiltration effluents. In the following years more lagoons were added and upgraded. In 1983, the network of lagoons and wetlands was declared a Ramsar site, internationally renowned for bird watching. The first modern lagoon was installed in 1986 (Hodgson and Paspaliaris 1996). Today all wastewater is treated through modern lagoons, and the last filtration paddocks were closed in 2004.

Nowadays, Western Treatment Plant actually comprises three WWTPs: the 25W, 55E, and 115E lagoon systems. Two of these plants—25W and 55E—receive raw wastewater, while 115E receives partially treated effluents from pond number 4 of 55E. Hence, only 25W and 55E currently operate a covered anaerobic pond, or "anaerobic pot," as it is referred to by the operator.

While 115E is a mere pond system, both 25W and 55E also include CAS systems for enhanced nutrient removal, commissioned in 2004 and 2001, respectively. These CAS plants each receive effluents from pond number 4 and then discharge their effluents into pond number 5 of their respective series of ponds (Melbourne Water 2014).

The key features of these plants are summarized in table CS4-2, and some pictures are provided in figures CS4-4 and CS4-5.

Table CS4-2: Key characteristics of WWTPs in Melbourne, analyzed for case study 4

WWTP	Technology	Total ponds		Anaerobic ponds		
		number (no.)	area (ha)	number (no.)	water depth (m)	area (ha)
Western Treatment Plant						
25W	AP - MAP - CAS - MP	11	264	1	2.9	8.0
55E	AP - MAP - CAS - MP	11	206	1	2.1	9.3
115E	MP	11	200	n.a.	n.a.	n.a.
Average		**11**	**223**	**1**	**2.5**	**8.7**
Total		**33**	**669**	**2**	**5.0**	**17.3**

Source: Melbourne Water.
Note: AP = anaerobic pond; CAS = conventional activated sludge; MAP = mechanically aerated pond; MP = maturation pond.
n.a. = not available.

Figure CS4-4: Western Treatment Plant, Melbourne

Source: Google Earth.

Figure CS4-5: Covered APs at Western Treatment Plant, Melbourne

Source: www.melbournewater.com.au.

For details of actual wastewater characteristics and other parameters, see tables CS4-3 and CS4-4. For details regarding general design recommendations for APs, see table CS4-12.

4.1.3. Sludge management

Santa Cruz, Bolivia

The sludge from anaerobic ponds has never been removed to date (SAGUAPAC 2014b). This is an astonishing feature, since the covers were already installed in 2006–7, and a typical sludge removal interval for APs would be one to three years, at most. Apparently, the two-millimeter sieves installed prior to the AP are having a positive impact on sludge removal intervals.

Melbourne, Australia

Sludge originates from two sources (Melbourne Water 2014):

- Anaerobically digested primary sludge from the 25W and 55E anaerobic ponds, including also settled grit and screenings
- Waste activated sludge (WAS) from the CAS systems

Removed sludge is discharged into twenty-four sludge drying pans (drying area = two hectares each), loaded at 400 tons DS/ha. The consequent sludge drying capacity equals 19,200 tons DS/y. This is just about 50 percent of the sludge production. Therefore, a significant proportion of sludge accumulates in the ponds. Work is currently underway to remove sludge accumulated in lagoons and implement a new sludge management strategy (refurbishment of old/disused drying pans, to enable loading of 750 tons DS/ha in these drying pans) that will increase the sludge drying capacity to 48,000 tons DS/y. During drying, not only is water evaporation taking place, but about 20 percent of DS is also destroyed.

After drying, the sludge is stored in stockpiles. These are almost at capacity and will be reconfigured to contain an additional five years' worth of solids production.

Research into beneficial reuse of sludge is ongoing.

For details of actual sludge characteristics, see table CS4-4.

4.1.4. Energy management

Santa Cruz, Bolivia

While the covers were already installed in 2006–7, the flaring of biogas was installed later and only went into operation in June 2009. For each of the two sites, there is a separate flaring station; these are called AT500 for System North and AT1000 for System East.

The only problems that have been observed are (a) inefficient condensate removal, which is particularly important due to the long biogas pipes to the flare, and (b) air intrusion into the biogas system. Since gas is collected by a suction system, the problem is not so much the loss of biogas, but the reduced methane content in the gas. The problem was solved by thermo-welding of small fissures in the plastic cover, implemented by a local company.

At present, a project is in the pipeline to install a CHP facility at each of the two sites to produce electric power.

Each of the systems actually available, both related to AT500 and AT1000, consists of the following (Libhaber and Orozco-Jaramillo 2012; Libhaber 2010):

- Cover, made from HDPE, 1.5 mm, UV resistant, 0.94 g/cm3
- Floaters, counterweights, and inspection openings
- Foam trap
- Condensate removal
- Suction compressors
- Scrubbing tower (only for industrial WWTP, for H_2S removal)
- Flow meter
- Flare
- Pipes HDPE, valves, and so on, as required
- Control system

The biogas utilization project in the pipeline foresees the following (Ghetti 2013):

- Co-generation:
  - AT500: two co-generators with 280 $kW_{electric}$ each (total = 560 $kW_{electric}$)
  - AT1000: five co-generators with 280 $kW_{electric}$ each (total = 1400 $kW_{electric}$)
  - TOTAL = 1,960 $kW_{electric}$
- Electric installations; transformer for supply to public grid
- Gas scrubber

For details of actual biogas characteristics, see table CS4-5.

Melbourne, Australia

Biogas from the 25W and 55E anaerobic ponds is sent to a power generation plant owned and operated by the local energy company, AGL.

About 99 percent of the electricity generated is used to operate the WWTP. The ratio between the plant's own production and purchased electric power is about 3:1. The biogas thus supplies about 75 percent of the total WWTP's electricity. Just a tiny 1 percent is also supplied to the public grid.

In fiscal year 2012–13, more than 10 percent of the produced biogas was still flared. This is attributed to a power station that is sized to maximize return on investment. Hence, particularly in summer when biogas production is above average due to elevated temperatures, some gas cannot be utilized and is flared.

Table CS4-13 summarizes general design recommendations for the biogas/energy component of covered APs.

4.2. Analysis

4.2.1. Wastewater influent, effluent, and other parameters of interest

Table CS4-3 provides key flow and load characteristics of the Santa Cruz WWTPs and Melbourne Western WWTP. Table CS4-4 provides information on actual wastewater influent and effluent quality and sludge production. All numbers in both tables are based on private communications received from the WWTP operators and relate to the year 2013. The presented quality data on Santa Cruz's four facilities are flow weighted, since those four plants treat different catchments with slightly different wastewater qualities.

Table CS4-3: Key general characteristics of Santa Cruz WWTP and Melbourne Western WWTP, average data from the year 2013

			Sta. Cruz	Melbourne
GENERAL				
Daily flow rate	average	m^3/d	118,000	482,000
	max. month		127,000	560,000
BOD_5 load	average	$kgBOD_5/d$	48,000	274,000
	max. month		61,000	300,000

Sources: SAGUAPAC and Melbourne Water—private communications 2014.

Table CS4-4: Actual influent and effluent data and sludge production of Santa Cruz WWTP and Melbourne Western WWTP, average data from the year 2013

			Sta. Cruz WWTP	Melbourne WWTP
Number of WWTPs			4	2
Pop. equivalents	avg. actual	PE_{60}	802,000	4,569,000
	max. month	PE_{60}	1,023,000	4,994,000
WASTEWATER QUANTITY				
Specific wastewater production		$m^3/PE_{60}/y$	54	39
		$L/PE_{60}/d$	147	105
WASTEWATER QUALITY				
COD	Influent	mg/L	946	1,009
	Effluent	mg/L	197	32
	Elimination	%	79	97
BOD_5	Influent	mg/L	407	571
	Effluent	mg/L	60	4
	Elimination	%	85	99
N_{total}	Influent	mg/L	92	73
	Effluent	mg/L	66	21
	Elimination	%	28	72
NH_4-N	Effluent	mg/L	2	5
NO_3-N	Effluent	mg/L	0	15
P_{total}	Influent	mg/L	15.2	10.5
	Effluent	mg/L	4.4	9.0
	Elimination	%	71	14
SLUDGE PRODUCTION				
Total annual DS production (tons DS/y)			n.a.	37,000
DS content at removal from ponds (% DS)			n.a.	6.55
Specific sludge volume produced ($L/PE_{60}/y$)			n.a.	124
Specific DS load produced ($gDS/PE_{60}/d$)			n.a.	22

Sources: SAGUAPAC and Melbourne Water, private communications 2014.
Note: The Melbourne effluent values are from 25W and 55E.
1 cap = 40 g BOD_5/d in Bolivia and 60 g BOD_5/d in Australia;
1 PE_{60} = 1.50 PE_{40} = 1.50 cap in Bolivia and 1.0 cap in Australia;
$m^3/PE_{60}/y$ x 1.50 = m^3/cap/y in Bolivia, and $m^3/PE_{60}/y$ x 1.0 = m^3/cap/y in Australia;
$L/PE_{60}/d$ x 1.50 = L/cap/d in Bolivia, and $L/PE_{60}/d$ x 1.0 = L/cap/d in Australia.
n.a. = not available.

4.2.2. Biogas production and potential for energy generation

Biogas production

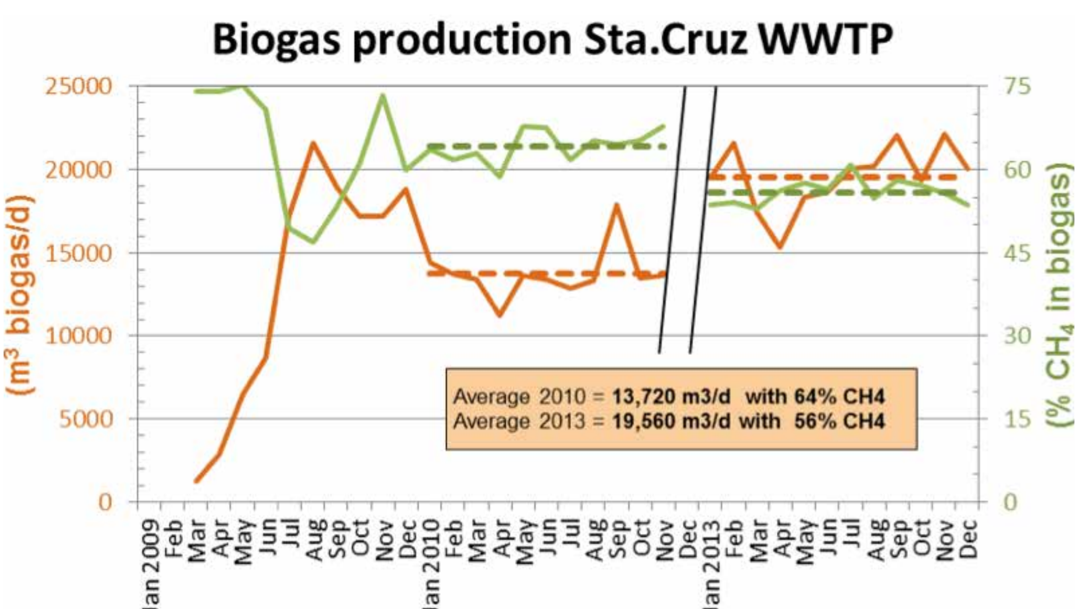
Table CS4-5 summarizes biogas data from both WWTPs of this case study. Furthermore, it contains some general parameters that are of interest for the assessment of the collected biogas. Figure CS4-6 presents Santa Cruz's annual biogas and CH_4 fluctuation during the years 2009, 2010, and 2013.

Table CS4-5: Biogas data of Santa Cruz WWTP and Melbourne Western WWTP

	Sta. Cruz		Melbourne
	2010	2013	2012–13
Average wastewater temp. (°C)	n.a.	27.0	20.8
Retention time in AP (d)	15	12	1
BOD_5 in raw wastewater (mg/L)	430	407	571
COD in raw wastewater (mg/L)	972	946	1,009
COD/BOD_5 (—)	2.3	2.3	1.8
COD removal efficiency (%)	63	n.a.	41
BIOGAS COLLECTION			
L/PE_{60}/d	20.6	24.6	13.1
L/$kgCOD_{destroyed}$	238	n.a.	297
CH_4-content (%)	64	56	79
METHANE COLLECTION			
L/PE_{60}/d	13.2	13.7	10.3
L/$kgCOD_{destroyed}$	153	n.a.	235

Source: Authors' calculation.
n.a. = not available.

Figure CS4-6: Daily biogas production and biogas CH_4 content at Santa Cruz WWTP



Source: SAGUAPAC.

Discussion:

- When covering anaerobic ponds, it takes several months until biogas collection reaches steady state conditions. In Santa Cruz it took half a year, from March to August 2009, to reach that point.
- Biogas quantity always should be considered together with its CH_4 content. While this holds generally true for all biogas systems, it appears that covered ponds are especially susceptible to changes in CH_4 content. The example from Santa Cruz demonstrates this quite clearly. It has been reported there that, due to small leaks, air is sucked into the biogas. Even though these leaks have been repaired, overall methane content to have fallen by about 20 percent, from 64 percent CH_4 in 2010 to 56 percent CH_4 in 2013.
- The increase of biogas quantities in Santa Cruz by 40 percent from 2010 to 2013 is thus explained as 20 percent caused by air intrusion and the remaining 20 percent attributed to an overall influent load increase of 20 percent in the same period.
- Furthermore, it is particularly important to distinguish between biogas production and biogas collection. As has been shown in case study 3 (UASB), a considerable portion of the produced biogas is lost, either dissolved in the liquid effluent or through imperfections in the collection system. Stoichiometrically, 1 $kgCOD_{destroyed}$ produces 350 L CH_4. The CH_4 collected is 153 L $CH_4/kgCOD_{destroyed}$ in Santa Cruz and 235 L $CH_4/kgCOD_{destroyed}$ in Melbourne. Hence, the losses observed at the WWTPs of case study 4 are 56 percent and 33 percent, respectively.
- An interesting feature is that the CH_4 yields for $L/PE_{60}/d$ and $L/kgCOD_{destroyed}$ point in different directions when comparing the two plants. While the former parameter presents Santa Cruz as the plant with higher CH_4 yield (13.7 versus 10.3 $L/PE_{60}/d$), the latter suggests Melbourne has a higher CH_4 yield (153 versus 235 $L/kgCOD_{destroyed}$). None of the various parameters analyzed (wastewater temperature, retention time in AP, BOD_5 and COD characteristics, or removal rates) provides an explanation.

Conclusions:

- For APs, it is recommended to prefer CH_4 yields over biogas yields, because intrusion of air can significantly change the total biogas quantities.
- A suitable estimate can be obtained interpolating a CH_4 yield of 10–14 $L/PE_{60}/d$ for average annual wastewater temperatures of 21–27°C.
- Alternatively, it is possible to use a CH_4 yield of 350 $L/kgCOD_{destroyed}$ and assume losses of 40 percent for concentrated wastewater with COD = 1000 mg/L. For thinner wastewater, the loss assumption should be increased according to dilution.

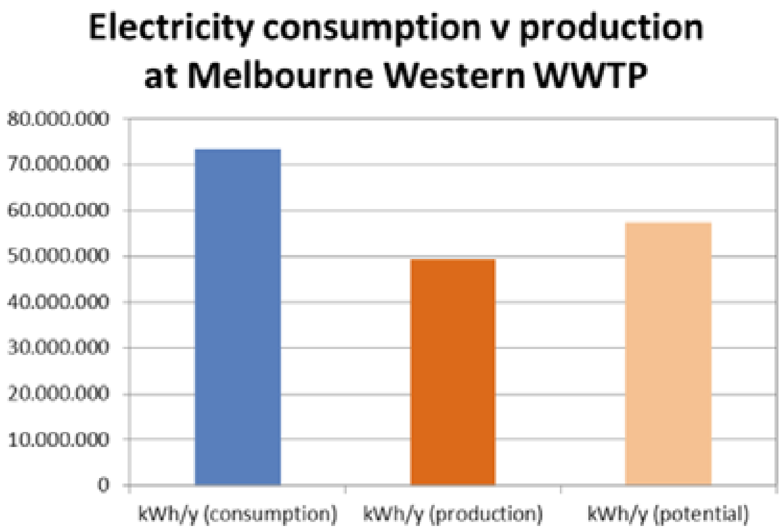
Potential for energy generation

Based on the above-recommended CH_4 yield of 10–14 $L/PE_{60}/d$, a calorific value of CH_4 of 10 kWh/m^3, and an electric CHP efficiency of 30–35 percent, the expected electricity generation potential of covered AP systems is about 11–18 $kWh/PE_{60}/d$.

At *Santa Cruz WWTP*, this electricity generation potential is definitely higher than the energy requirements for the existing pond system. No energy consumption data were available for this case study, yet wastewater pumping is the main electric power consumer. It is estimated that this requires about 0.5–4 $kWh/PE_{60}/y$, as demonstrated in figure CS3-7 for a wide range of scenarios. Consequently, it is important to find an attractive use for this surplus.

At *Melbourne Western WWTP*, the situation is different, since two energy-intensive CAS systems have to be operated in addition to the ponds. Figure CS4-7 presents the electricity balance for these conditions. (Note: The value for “kWh/y [potential]” includes the flared gas.)

Figure CS4-7: Power generation potential of actual biogas production, compared to actual electricity consumption at Melbourne Western WWTP in 2013



Source: Melbourne Water.
Note: Data refer to 2013.

Conclusions:

- The electricity generation potential from covered APs is about 11–18 kWh/PE_{60}/d.
- This potential is sufficient to cover the needs of wastewater systems with low energy consumption, such as WSP (for example, Santa Cruz WWTP) or TF (see case study 4).
- In cases of more energy-intensive treatment technologies, such as subsequent CAS systems, the electricity generation potential from covered APs is only sufficient for partial electricity coverage, on the order of approximately 50–70 percent (for example, Melbourne Western WWTP). Note that the energy consumption of this combination of covered AP + CAS requires less electric power than conventional CAS systems with PST (enhanced mechanical carbon removal, no separate sludge digester).

Table CS4-6 presents a summary of key biogas and power generation parameters.

Table CS4-6: Biogas and power generation potential of covered AP at Santa Cruz WWTP and Melbourne Western WWTP

	Sta. Cruz	Melbourne
Biogas production		
- N elimination (L/PE$_{60}$/d)	**—**	**—**
- C elimination (L/PE$_{60}$/d)	**24.6**	**13.1**
Electric efficiency CHP (%)	30	35
Thermal efficiency CHP (%)	50	50
Calorific value of biogas (kWh/m^3)	5.6	7.5
Power generation		
- N elimination (kWh/PE$_{60}$/year)	**—**	**—**
- C elimination (kWh/PE$_{60}$/year)	**15.1**	**13.2**
Thermal energy generation		
- N elimination (kWh/PE$_{60}$/year)	—	—
- C elimination (kWh/PE$_{60}$/year)	25.1	18.8
Total energy generation		
- N elimination (kWh/PE$_{60}$/year)	—	—
- C elimination (kWh/PE$_{60}$/year)	40.1	32.0

Source: Authors' calculation.

Note: L/PE$_{60}$/d x 16.67 = L/kg BOD$_5$/d; kWh/PE$_{60}$/y x 16.67 = kWh/kg BOD$_5$/y. The biogas and electricity potential of Melbourne WWTP is assigned to "C elimination," since it is not connected to the N removal of the downstream CAS system.

4.2.3. Operation capacity needs, biogas safety

The key operational and safety issues are considered similar to those of UASB reactors: safety concerns, deposits in the reactor, insufficient biogas treatment, and scum formation. Hence, for details see case study 3, section 3.2.3.

4.2.4. Institutional aspects, energy costs

Bolivia

Any physical or legal entity has the right to produce and consume its own electric energy in Bolivia. As long as the installed electric power does not exceed 2000 kW, doing so requires only registration with the "Autoridad de Fiscalización." If the installed power exceeds 2000 kW, the applicant requires a power generation license, which is more complicated and costly to obtain. Given these legal boundaries and the quantities of biogas involved at Santa Cruz WWTP, it was proposed by a study to install an electric power generation capacity of 1,960 kW (Ghetti 2013).

The electricity potential of the collected biogas is larger than the consumption at the WWTPs in question. Therefore, SAGUAPAC intends to utilize the surplus electricity in water supply, which also falls under its responsibilities. To supply the autogenerated electric power from the two WWTPs to the water supply sites requires using the public grid for transportation. Even though it has never been done in Bolivia before, it is legally possible to pay the electrical distribution utility for the use of its network to transport SAGUAPAC`s electricity from its generation point (the anaerobic lagoons) to other consumption points (water treatment and pumping systems). In other words, SAGUAPAC would not sell the electricity it produces to the electrical utility; it would pay for its transportation. The electricity generation project currently in the pipeline thus provides for all necessary installations for that purpose and includes an estimate for the cost of utilizing the public grid in its financial assessments (Ghetti 2013). For results, see section 4.2.8.

The electricity tariff in Bolivia is different at different times of day and features various surcharges. Currently, the effective unit cost for electric power is US$0.065/kWh (EUR0.048/kWh), as cited by SAGUAPAC (2014b). The assumed future electricity cost savings are based on a tariff of US$0.085/kWh (EUR0.063/kWh), since a considerable portion of the electricity production from biogas is assumed during periods when the unit cost is peaking (Ghetti 2013).

Australia

The priority in Australia is on utilization of the generated electricity onsite for the operation of the WWTPs. Just a tiny 1 percent of the electricity production is supplied into the public grid. This is understood to be a consequence of issues with balancing power production with supply.

Generally, the typical average unit cost paid by Melbourne Water is about **US$0.09/kWh** (EUR0.067/kWh).

4.2.5. GHG reduction and CDM co-financing

General

For general information on CDM co-financing, the price of carbon credits, and related matters, see section 1.2.5. Also interesting is the question of how anaerobic wastewater treatment compares to aerobic wastewater treatment in terms of GHG emissions; for details, see section 3.2.5.

Specifics of case study 4

The "additionality" criterion (see section 1.2.5) means that new construction of a covered anaerobic pond plus biogas collection and utilization are not eligible under CDM for their methane mitigation effects. In cases like this one, in which the project both creates and eliminates part of the methane, only the energy production component from a renewable source that replaces fossil sources may be considered for CDM on the plus side, while the additional CH_4 emissions generated by the project, but lost dissolved in the pond effluents, have to be considered on the minus side.

The CDM situation is different for Bolivia, where most existing ponds are not covered (baseline scenario) and thereby produce methane emissions to the atmosphere. In the case of Bolivia, CDM carbon credits would apply both to the methane mitigation and the renewable energy generation.

The country-specific energy mixtures shown in table CS4-7 derive from the GHG emissions for electricity generation in Australia and Bolivia.

Table CS4-7: GHG emissions per kWh for electricity generation in Bolivia and Australia, as compared to the world

Region	1990 gCO_2/kWh	2010 gCO_2/kWh
Bolivia	307	**423**
Australia	817	**841**
World	586	**565**

Source: IEA 2012.

Note: The table shows CO_2 emissions from fossil fuels consumed for electricity generation, in both electricity-only and combined heat and power plants, divided by output of electricity generated from fossil fuels, nuclear, hydro (excluding pumped storage), geothermal, solar, wind, tide, wave, ocean, and biofuels. Both main activity producers and autoproducers have been included in the calculation.

Table CS4-8 provides a quantitative assessment of the CO_{2e} emission reduction, assuming both the renewable electricity generation in CHP and the CH_4 emission reduction would apply to either WWTP of this case study. The numbers thus highlight roughly which orders of magnitude to expect for GHG emission reductions from these installations.

Table CS4-8: GHG emission reduction due to electric power generation and CH_4 mitigation at Santa Cruz WWTP and Melbourne Western WWTP

	Santa Cruz	Melbourne
Electricity potential or production in CHP (kWh/y)	11,970,000	49,456,000
Substituted fossil CO_2 emissions (gCO_2/kWh)	423	841
GHG reduction by electric power generation (tons CO_{2e}/y)	**5,063**	**41,592**
Methane emission reduction through flare and CHP (m^3 methane/ year)	3,990,000	17,224,000
Specific weight of methane (kg/m^3 methane)	0.72	0.72
GHG factor of methane (—)	21	21
GHG reduction by CH_4 emission reduction (tons CO_{2e}/y)	**60,329**	**260,427**
TOTAL GHG reduction (tons CO_{2e}/y)	**65,392**	**302,019**

Source: Authors' calculation.

4.2.6. CAPEX structure

Santa Cruz

The CAPEX estimate for the biogas utilization project in Santa Cruz is summarized in table CS4-9.

This summary contains both all investments made to date in the context of the biogas recovery project (SAGUAPAC 2014a) and the investments still planned for CHP (Ghetti 2013).

Table CS4-9: CAPEX for biogas project at Santa Cruz WWTP

	CAPEX US$
IMPLEMENTED	
Influent sieves 2 mm	822,336
Civil works for sieves	221,427
Covers on APs	1,733,275
Biogas collection system	312,558
Biogas flaring system	501,996
Civil works for gas flaring system	11,050
Electricity supply to flares	48,742
Low energy mixers/aerators downstream of APs	1,147,790
Other	12,000
SUBTOTAL 1	3,663,384
PLANNED	
CHP: 7 x 280 kW electric	1,286,447
Electric installations and transformers	300,320
Civil works	60,000
Gas filters	37,577
Installation works	21,000
Contingencies	170,534
SUBTOTAL 2	1,875,878
TOTAL	**5,539,262**

Sources: SAGUAPAC 2014a; Ghetti 2013.
Note: The cost for the covers of US$1.73 million, combined with the total covered pond area of 12.3 ha, leads to a unit cover cost of about US$14.1/m2.

Melbourne

No CAPEX information available.

4.2.7. OPEX structure

Santa Cruz

The latest OPEX estimate for the biogas utilization project in Santa Cruz is summarized in table CS4-10. This summary contains both the OPEX increase due to the investments made to date in the context of the biogas recovery project (SAGUAPAC 2014a) and the additional OPEX expected once the planned CHP is installed, taking into account that the surplus power will be transferred through the public grid to other SAGUAPAC facilities consuming electricity (Ghetti 2013).

Table CS4-10: OPEX for biogas project at Santa Cruz WWTP

	OPEX US$/y
IMPLEMENTED	
Electric energy	5,000
Maintenance of equipment	6,000
Maintenance of covers	49,000
SUBTOTAL 1	60,000
PLANNED	
CHP: O&M	131,057
Additional manpower, insurance, administration	65,809
Fee for using the public grid	51,129
Electricity savings	-897,930
SUBTOTAL 2	-649,935
TOTAL	**-589,935**

Sources: SAGUAPAC 2014a; Ghetti 2013.

Consequently, while to date OPEX has increased by US$100,000 per year, after the installation of CHP it will be reduced by about US$590,000 per year once it is working.

Melbourne

No OPEX information available.

4.2.8. Viability of investment in biogas utilization

Santa Cruz

Table CS4-11 summarizes several cost indicators from case study 4.

Table CS4-11: Cost indicators for biogas utilization project at Santa Cruz WWTP

Average influent load	$PE_{60,avg}$	802,000
Peak influent load	$PE_{60,max}$	1,023,000
CAPEX	US$	5,539,262
OPEX	US$/y	-589,935
specific CAPEX	US$/$PE_{60,avg}$	6.91
	US$/$PE_{60,max}$	5.41
specific OPEX	US$/$PE_{60,avg}$	-0.74
	US$/$PE_{60,max}$	-0.58

Source: Authors' calculation.

Note: 1 PE_{60} = 1.50 PE_{40} = 1.50 cap in Bolivia; US$/$PE_{60}$ x 16.67 = US$/kg BOD_5.

Overall, without interest, total CAPEX is covered through reduced OPEX (simple payback) in about ten years. The main reason for the long return period is the low cost of electricity of US$0.085/KWh.

However, one could also argue that the covers were necessary anyhow for bad odor elimination. This approach was taken in Ghetti (2013), which only looked at CAPEX and OPEX for CHP and rendered the project as financially very viable. Hence, under the given circumstances, there is a clear incentive for SAGUAPAC to proceed with the implementation of the CHP component.

Melbourne

No assessment of financial viability has been possible for Melbourne, since no CAPEX or OPEX information was disclosed by the operator.

4.3. Conclusions for covered ponds in EAP countries

The key characteristics of anaerobic ponds (APs) described in case study 4, such as BOD_5 elimination, VS destruction, sludge characteristics, and biogas production, can be transferred from Australia and Bolivia to EAP countries without major changes. Based on similar temperature conditions, similar project-specific results can be attained.

A possible change relates to wastewater treatment requirements. The AP systems in Santa Cruz, Bolivia, were designed for carbon removal only; the ones in Melbourne were later upgraded with nutrient removal CAS stages downstream of AP. If, in EAP, additional nitrification is required, more efficient polishing stages than just ponds will be needed. This will generally push up a WWTP's CAPEX and increase its OPEX, particularly where CAS is applied for nitrification. For polishing trickling filters (TFs), OPEX will stay the same, if additional nitrification is required. After all, the pumping head does not change when more TFs need to be supplied to facilitate nitrification.

If denitrification is required as well, the additional installations will, in any case, increase CAPEX, OPEX, and energy consumption further. This might eventually lead to a situation where, with nitrification + denitrification, less than 100 percent of electricity consumption can be covered from biogas, even at more efficient, large plants. The Melbourne case study already shows that, if combined with CAS, about 50–70 percent of the overall energy consumption can be covered by the electricity from biogas. The Melbourne case also demonstrates a general problem: lack of carbon sources for denitrification after AP (how best to solve this issue is still under consideration). This is not to say that the AP and denitrification would not work, but denitrification downstream of such efficient BOD removal is generally challenging.

It is questionable, though, if the same amounts of biogas can be collected when wastewater is diluted, as is common in EAP. Due to the higher flow rate per capita, a higher percentage of the generated methane can be lost dissolved via the liquid effluents, even if the same quantities of biogas are generated. This will have a negative impact on the electricity generation potential, and it will increase the GHG emissions. It is also possible to assume a collectable biogas potential and, thus, electricity generation potential lower than the values indicated in case study 4, table CS4-6.

Dilution of wastewater also has negative impacts on the design of anaerobic ponds. One of the main design parameters is retention time in those ponds. Increasing flow rates thus lead to increasingly large (and more expensive) ponds.

The quantities of digested sludge will be lower in EAP than in Bolivia and Australia. This is particularly expected where septic tanks continue to be used in large numbers. CAPEX is expected to be similar in EAP to that in Bolivia, whereas it is expected to be higher in Australia. OPEX savings depend mainly on electricity unit cost and may run anywhere from lower to higher in EAP than in this case study.

Success stories in EAP

In general, there are not many examples of covered APs worldwide. The two examples from case study 4, in Melbourne (Australia) and Santa Cruz (Bolivia), are the best known plants of that type in Australia and Latin America, respectively.

There have also been some reports on covered APs for piggery waste from New Zealand and on facilities in Vietnam: at two WWTPs in Da Nang (Phu Loc WWTP and Ngu Hanh Son WWTP), existing anaerobic ponds were covered as a result of public concern over odor (World Bank 2013f). No biogas is being collected, but interest exists to explore the potential for wastewater-to-energy options.

COVERED ANAEROBIC PONDS: TECHNICAL SUMMARY

Table CS4-12 General design recommendations for covered anaerobic ponds

AP DESIGN

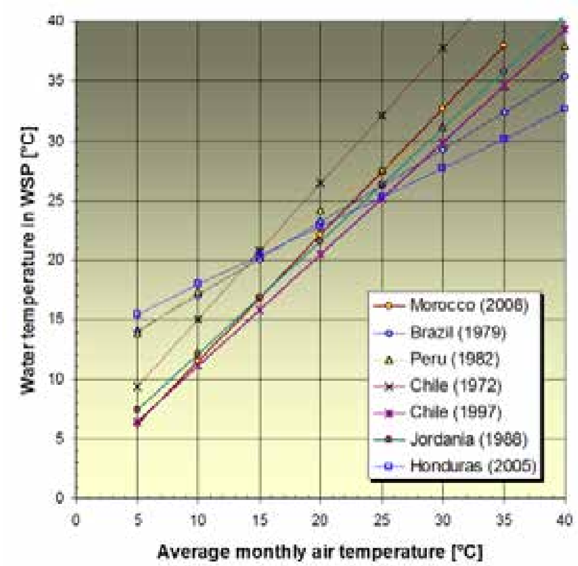
Retention time = >1 d, <5 d

- Volumetric BOD load
  - <10°C: <100 $gBOD_5/m^3/d$
  - 10–20°C: <20 x T(°C) – 100
  - 20–25°C: <10 x T(°C) + 100
- Sludge production:
  - 50–130 L/cap/y
  - 4–10 kgTS/cap/y
- BOD_5 removal efficiency:
  - <10°C: 40%
  - 10-25°C: 2 x T(°C) + 20
- COD effluent concentration: COD ≈ BOD_5 x 2.5
- SS removal efficiency: 60–80 percent
- N removal efficiency: 5 percent
- P removal efficiency: 5 percent
- FC effluent concentration: $FC_{eff} = FC_{inf} / (1 + k_T \times t)$, with

$k_T = 2.6 \times 1.15^{(T-20)}$, T: wastewater temperature (°C), t: hydraulic retention time (d)

Sources: Based on FWT 2008; Sperling and Chernicharo 2005; Mara and Pearson 1998.

Figure CS4-8: Prevailing wastewater design temperature of ponds as function of air temperature



Source: FWT 2008.

The prevailing wastewater temperature can thus usually be assumed to be about 1–2°C higher than the coldest average monthly air temperature.

AP SHAPE AND CONSTRUCTION

- Individual ponds should not be larger than 3 ha, to minimize wind impact.
- Length:width (at water surface) = 2:1, or larger
- Water depth = 2.0–5.0 m
- Internal embankment slopes with 1:2–1:3, external embankments with 1:1.5–1:2
- Minimum freeboard: 0.5 m (pond <1 ha), 0.5–1.0 m (pond with 1–3 ha)
- Positioning: flow direction counter-current to dominant wind direction
- If possible, install at least two parallel ponds for revision, and so on and bypass.
- Inlet: always below water surface and above sludge level—for example, at mid-water depth
- Outlet: scum wall 30 cm submerged into the water
- Outlet flow velocity: <0.1 m/s at the most unfavorable point
- Positioning: inlet and outlet in diagonally opposite corners of the pond
- Bottom lining: clay, or plastic sheet (mostly HDPE, PVC)

SLUDGE REMOVAL	• Sludge removal: mostly done with heavy machinery or with slurry pumps • Should be considered prior to construction
PLASTIC COVER	• Specialized companies for production and installation • Material: mostly HDPE, 1.5–2.5 mm, but also foam layer in the middle of two outer plastic layers (XR-5); polypropylene (PP) has been used. • UV light resistance • Anchored around the pond perimeter • Weighted pipes and so forth are used to hold the cover down. • Biogas migration below the cover must be possible. • Consider water level fluctuations in the pond. • Stormwater has to be drained safely. Some systems collect and remove the stormwater; others drain it into the pond water. • Typical lifespan: ffi 20 years.

Sources: Based on Libhaber and Orozco-Jaramillo 2012; Heubeck and Craggs 2009; NIWA 2008; DeGarie et al. 2000; Melbourne Water (www.melbournewater.com.au).

Table CS4-13: Covered anaerobic ponds: General design parameters and key characteristics of energy management

BIOGAS DESIGN	• *Biogas quantity* General: Few design references exist on biogas production in covered anaerobic ponds. The following approaches are mostly used for estimating the biogas potential in covered APs for municipal wastewater: (i) Biogas yield from influent BOD_5 load: Daily biogas production (m^3/d) = 0.3 x $kgBOD_{5,influent}/d$ (Libhaber and Orozco-Jaramillo 2012). The result from this calculation may be considered the collectable biogas quantity. Note: The above 300 L gas/$kgBOD_5$ is equivalent to 150 L gas/kgCOD; assuming 60 percent COD destruction in the AP, this is equivalent to 250 L gas/$kgCOD_{destroyed}$; based on 70 percent methane, this is 175 L $CH_4/kgCOD_{destroyed}$; compared to the theoretical maximum (see below) of 350 L CH_4/kg $COD_{destroyed}$, this implies a biogas loss assumption of 50 percent. (ii) Biogas from COD load destroyed in AP: Daily biogas production (m^3/d) = 0.350 x $kgCOD_{destroyed}/d$. This is a stoichiometric relationship. It describes how much biogas is produced, but it does not say anything about how much of this biogas can indeed be collected. After all, much biogas is lost dissolved in the effluent. With a loss assumption of 50 percent, the collectable daily biogas quantity thus is 0.175 x $kgCOD_{destroyed}/d$. (iii) Biogas from influent VS load: This approach is not recommended for municipal wastewater since it does not include the considerable dissolved influent organic load. It is more appropriate for industrial applications—for instance, in piggeries—where the bulk of the influent COD originates from VS load (Heubeck and Craggs 2009; NIWA 2008). Case study 4: The following has been found in this case study: (i) Biogas yield from influent BOD_5 load: Collected methane = 9.8–13.7 L $CH_4/PE_{60}/d$. That is equivalent to 160–230 L $CH_4/kgBOD_5$. Based on an average CH_4 content of 70 percent follows: Daily biogas production (m^3/d) = (0.23 to 0.33) x $kgBOD_{5,influent}/d$. (ii) Biogas from COD load destroyed in AP: Collected methane = 150–220 L/$kgCOD_{destroyed}$. Compared to the above cited maximum of 350 L/ $kgCOD_{destroyed}$, the losses thus amount to about 35–60 percent.

- *Biogas quality*

A typical CH_4 content from biogas of covered AP according to the literature is about 70 percent (Libhaber and Orozco-Jaramillo 2012). This case study found a range of 56–75 percent for CH_4 content.

BIOGAS TREATMENT *See table CS1-5.*
GAS HOLDER *(Not required; sufficient storage capacity below covers)*
FLARE *See table CS1-5.*
BIOGAS UTILIZATION *See table CS1-5.*

CASE STUDY 5: CO-DIGESTION OF ORGANIC WASTE

5.1. BACKGROUND, PROCESS DESCRIPTION

5.1. Background, process description

5.1.1. Data sources

Co-digestion of organic waste in sludge digesters has been practiced in hundreds of WWTPs for many years. In Europe, this concept has been used for more than three decades (WERF 2010b; Schmelz 2000). With the recent trend toward maximizing biogas yields and electricity production at WWTPs, this technology has gained rapid and ever more widespread popularity in developed countries. Indeed, it has evolved into the means most often used for substantially increasing the conventional biogas yield of sludge digesters. For instance, in the United States, both WERF (2012a) and Willis and Stone (2012) recommend the "use [of] alternative feedstocks to increase biogas production" as a key component in overcoming barriers to biogas use.

The traditional feedstocks used for co-digestion have been FOG (fats, oils, and grease) from fat traps from restaurants and canteens and food waste from food service establishments. Nowadays, municipal organic waste is also increasingly being used, as well as specific industrial wastes, particularly from food industries.

Case study 5 presents the experiences of a medium-sized municipal WWTP located in Zirl, Austria. It has a design capacity of 61,500 PE_{60} and 13,600 m^3/d (www.avzirl.at). Its startup was in 1996 as an extended aeration (EA) facility. In 2005 it was equipped with mesophilic digesters, and wastewater treatment switched to CAS. Co-digestion started soon after, in 2007, and has been practiced to date. The case study will present both initial results without co-digestion and subsequent results with co-digestion, making even more apparent the drastic changes of energy management due to co-digestion.

As long as mesophilic temperature conditions (30–38°C) are maintained in a digester and conventional design criteria are in place, the results from this case study, and from the additional references cited in this section, can be transferred to any other location worldwide.

5.1.2. Wastewater management

Since 2005, wastewater treatment at Zirl WWTP has been based on a classical CAS system, designed for enhanced P and N elimination, complete with preliminary treatment (screen, sand/fat removal), followed by primary sedimentation tank (PST), bioreactor, and secondary sedimentation tank (SST).

There was no need for a change of process technology in wastewater management due to the introduction of co-digestion. The only aspect to consider was the additional nitrogen (N) load in the filtrate from sludge dewatering, caused by the degradation of the extra organic feedstocks. Yet, as it turned out, these additional aeration requirements could be compensated for by optimizing the automated control of aeration in the bioreactors.

Figure CS5-1 presents two pictures from the plant. For details of actual wastewater characteristics and other parameters, see tables CS5-1 and CS5-2.

Figure CS5-1: Two different views of Zirl WWTP

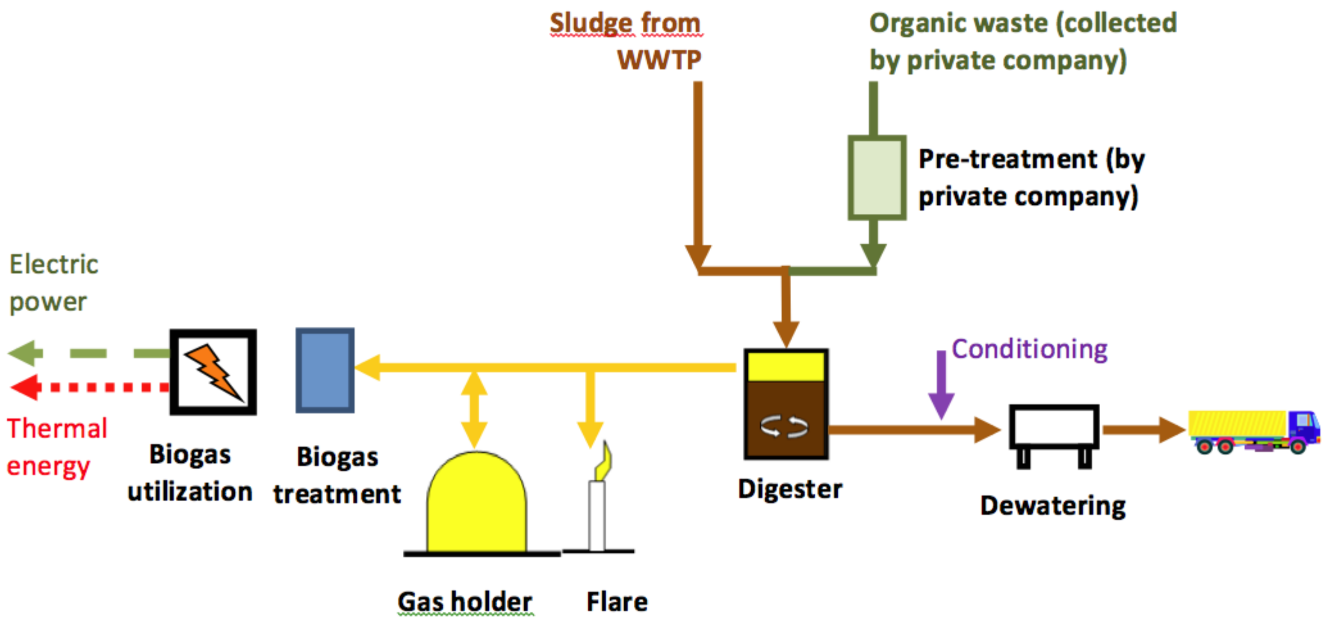
Source: Abwasserverband (AV) Zirl et al.

5.1.3. Sludge management

Sludge treatment at Zirl WWTP is based on a conventional concept, as presented in figure CS5-2: both primary and secondary sludge from wastewater treatment is thickened (gravimetrically and mechanically, respectively) and then digested in a mesophilic digester at 35°C (net volume = 1,350 m^3). The digested sludge is dewatered in a screw press and then disposed of by an external company. Both incineration and composting are currently applied to the digested and dewatered sludge. Biogas from digestion is balanced in a gas holder, treated, and finally utilized in CHP for the combined production of electric and thermal energy.

Organic waste that is collected by a private company is co-digested in the WWTP's single sludge digester. No relevant changes were made to the sludge/biogas train for that purpose. Only minor adjustments of existing installations to serve optimal takeover and feeding of organic waste were necessary.

Figure CS5-2: Simplified flow scheme of co-digestion at Zirl WWTP



Source: Authors.

Several types of organic waste are being or have been co-digested at Zirl WWTP:

- FOG (fats, oils, and grease)
- Leachate from composting (only from 2007 to 2010)
- Biowaste from an industrial bakery
- Biowaste from an industrial pizza producer
- Packed grocery stores' waste
- Municipal organic waste

Management of FOG and of all other organic feedstocks is done separately. The former is buffered in a heated holding tank to avoid solidification and then constantly fed to the digester. The FOG installations include mainly a FOG holidng tank of 10 m^3, completed with heating and grinding + feeding system to digester.

All other organic feedstocks (from bakery, pizza producer, grocery stores' waste, and municipal organic waste) undergo more intensive pretreatment. The following installations are available for that purpose:

- *Reception tank:* Trucks deliver the various types of organic waste and dump it into this tank. From here a spiral conveyor transports the waste to a hammer mill.
- *Hammer mill:* Manufactured by Wackerbauer, Germany, this mill is used for crushing coarse organic material and reducing it in size and removing unwanted materials (in this case, plastics, wood, stones, and metals, in particular). Liquors are added as necessary to obtain a suspension that can be easily pumped. Practice has shown that for the wastes in question, suspensions with a maximum DS of about 15–20 percent are still manageable. The liquors used for slurrying are typically of organic waste origin, as

well—for example, leachate from composting, and/or whey. The maximum grain size after the hammer mill usually is 8–10 mm.

- In this specific case, the reception tank and hammer mill are located not at the WWTP site, but at the premises of a nearby private company that collects the waste for co-digestion. The treated product is then pumped to a suction truck and transported to the WWTP, where it is unloaded into the biowaste holding tank. See figure CS5-3 for the hammer mill and unloading of pretreated organic waste at Zirl WWTP. Figure CS5-4 shows typical characteristics of pretreated food waste.

- *Biowaste holding tank:* Located at the WWTP, 110 m3, complete with mixing and feeding system to the digester

Figure CS5-3: Hammer mill and organic waste unloading at Zirl WWTP

Source: Abwasserverband (AV) Zirl et al.

Figure CS5-4: Liquefied food waste prior to co-digestion at Zirl WWTP

Source: Abwasserverband (AV) Zirl et al.

The combined thickened primary and secondary sludge from wastewater treatment and the pretreated organic wastes are co-digested in a single closed digester (1,350 m^3) at mesophilic temperatures of about 35°C. The digester is of cylindrical type (see figure CS1-22), with diameter:sludge depth ≈ 13.0:12.6 m. It is constructed out of concrete, featuring thermal insulation and external heat exchangers for temperature control. The digester is equipped with biogas injection for mixing. Digested sludge is conditioned with polymers, dewatered in a screw press to about 25 percent DS, and then transported to composting or to an incineration facility for thermal reuse.

For details of actual sludge and feedstock characteristics, see tables CS5-2 and CS5-3.

For details on general design parameters and key characteristics of anaerobic digesters, see table CS1-4. Specific impacts of co-digestion on sludge management are summarized in table CS5-7.

5.1.4. Energy management

At Zirl WWTP, the biogas produced in the digester is balanced in a gas holder, treated, and finally utilized in CHP for the combined production of electric and thermal energy. Almost no gas is flared. The main elements of biogas management are the following:

- Foam trap
- Condensate removal
- Two-stage activated carbon filter for H_2S and siloxane removal
- Gas holder (400 m3)
- Flare
- CHPs: (a) co-generation no. 1 with 105 kW electric power, 165 kW thermal power; (b) co-generation no. 2 with 75 kW electric power, 145 kW thermal power; total electric power = 180 kW. Electric efficiency of no. 1 is 34.7 percent and of the older no. 2 just 25 percent.
- Pipes, valves, and so on as required

The electricity produced from biogas is primarily used for the WWTP's own energy requirements. Only a small percentage of the generated electric power is supplied into the public grid when production is above consumption levels.

Co-digestion at Zirl WWTP has not changed biogas characteristics. Typically, the calorific value of the gas was and continues to be about 6.2 kWh/m^3—that is, the biogas has 62 percent CH_4 content.

For details on general design parameters and key characteristics of biogas systems combined with anaerobic digesters, see table CS1-5. For general design parameters relevant to energy management with co-digestion, see table CS5-9.

5.2. Analysis

5.2.1. Wastewater influent, effluent, and other parameters of interest

Wastewater

Table CS5-1 provides key flow and load characteristics of Zirl WWTP. Table CS5-2 provides actual information on wastewater influent and effluent quality. All numbers in both tables relate to the year 2012.

Table CS5-1: Key general characteristics of Zirl WWTP, average data from the year 2012

			Actual	Design
GENERAL				
Daily flow rate	average	m^3/d	9,616	—
	85 percentile		12,700	13,600
Pop. equivalents	average	PE_{60}	42,950	—
	85 percentile		54,200	61,500

Source: Abwasserverband (AV) Zirl et al.

Table CS5-2: Actual influent and effluent data and sludge production of Zirl WWTP, average data from the year 2012

			Zirl WWTP
Number of WWTPs			1
Pop. equivalents	avg. actual	PE_{60}	42,950
	85 percentile	PE_{60}	54,200
WASTEWATER QUANTITY			
Specific wastewater production		$m^3/PE_{60}/y$	82
		$L/PE_{60}/d$	224
WASTEWATER QUALITY			
COD	Influent	mg/L	504
	Effluent	mg/L	29
	Elimination	%	94
BOD_5	Influent	mg/L	282
	Effluent	mg/L	5
	Elimination	%	98
N_{total}	Influent	mg/L	35.0
	Effluent	mg/L	10.7
	Elimination	%	69
NH_4-N	Effluent	mg/L	2.8
NO_3-N	Effluent	mg/L	6.5
P_{total}	Influent	mg/L	6.5
	Effluent	mg/L	0.6
	Elimination	%	91
SLUDGE PRODUCTION			
Total annual sludge production (tons dewatered sludge/y)			2,322
DS content after dewatering (percent DS)			25
Specific DS load produced ($gDS/PE_{60}/d$)			37

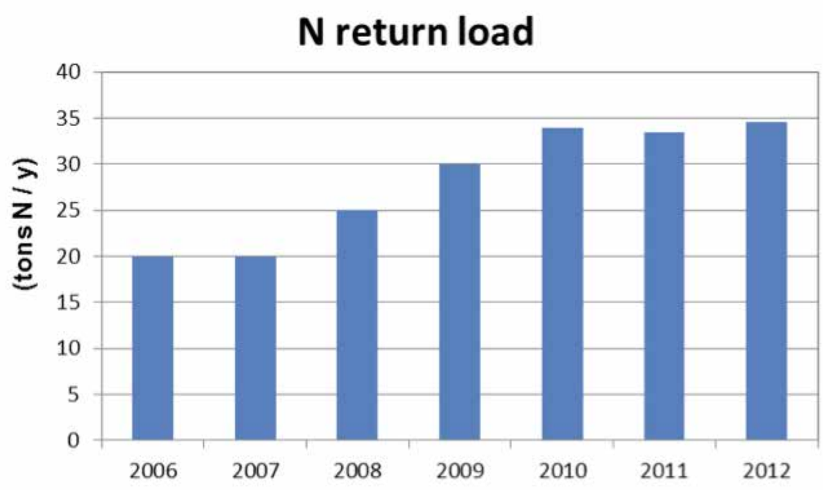
Source: Abwasserverband (AV) Zirl et al.

Note: 1 cap = 60 g BOD_5/d in Austria; 1 PE_{60} = 1.0 cap in Austria; $m^3/PE_{60}/y$ = m^3/cap/y in Austria; $L/PE_{60}/d$ = L/cap/d in Austria.

Return load from dewatering filtrate

An important question regards the **additional nitrogen (N) return load** via filtrate from sludge dewatering. Figure CS5-5 presents the total annual N return load registered at Zirl WWTP.

Figure CS5-5: N return load in filtrate from sludge dewatering at Zirl WWTP



Source: Abwasserverband (AV) Zirl et al.

Apparently, the N return load increased from about twenty to thirty-three tons N/year between 2007 and 2012. As a percentage of influent N load, the N return load increased from about 17 percent to 30 percent. Relatively speaking, N return load thus increased by 75 percent at Zirl WWTP. This is a rather large increase, the size of which has not been found in all the case studies. Svardal and Haider (2010) report on five WWTPs in the same region where co-digestion led to a wide variation of impacts on N return load, ranging from no impact to a maximum increase of 14 percent, whereas Nowak and Ebner (2013) report on a case study where return N load increased by more than 100 percent. In this last case, all raw co-digestates featured a high protein and nitrogen content.

In conclusion, generalizations are not possible. The additional N load always needs a case-specific assessment. Thus, the impact on aeration requirements in secondary wastewater treatment also has to be assessed on a case by case basis.

At Zirl WWTP, there has been an increase in N return load, which was compensated for in practice by an improved aeration control and automation system. Notwithstanding, for fair comparison, the cost impact of this increased N return load must be considered and is part of the OPEX assessment in section 5.2.7.

Sludge

The main types of feedstock to the digester at Zirl WWTP are the following:

- Sludge: Primary sludge (from gravity thickening after primary sedimentation) + WAS (from mechanical thickening after secondary wastewater treatment)
- Waste: FOG + organic waste mixture + leachate from composting

Table CS5-3 presents the typical characteristics of these feedstocks.

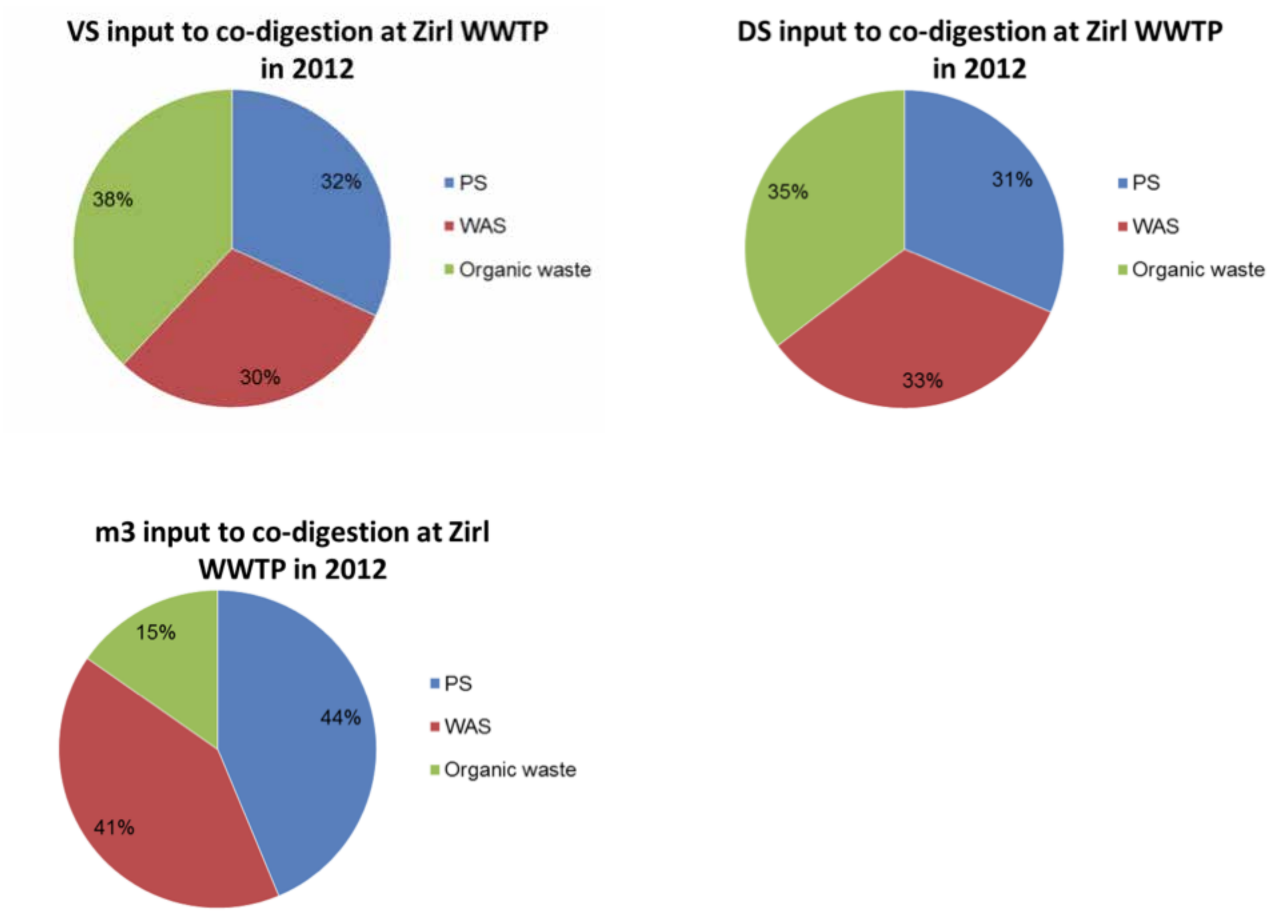
Table CS5-3: Typical characteristics of feedstocks used at Zirl WWTP

	SLUDGE		WASTE		
	Primary sludge	WAS	FOG	Organic waste mixture	Leachate from composting
DS (%)	5.4	6.1	6.0	18.3	5.0
VS (%)	4.4	4.4	5.7	15.5	2.5
COD (mg/L)	85,000	62,000	127,000	291,000	74,000
COD/VS (—)	1.9	1.4	2.3	1.9	2.9

Source: Abwasserverband (AV) Zirl et al.

In terms of the distribution of VS and DS load added to the digester, sludge (PS + WAS) makes up about two-thirds and organic waste is responsible for about one-third of total input. In terms of volume added, the sludge represents 85 percent of loading. Figure CS5-6 shows the average VS, DS, and volume distribution for the year 2012.

Figure CS5-6: Distribution of VS, DS, and volume input to co-digestion at Zirl WWTP in 2012



Source: Abwasserverband (AV) Zirl et al.

The additional waste feedstock produced an increase in dewatered sludge quantities by about 10 percent, which was attributed to the remains of waste feedstock after digestion. This increase is less than what should be expected from calculations of sludge production, according to table CS5-7. Apparently, the mutual digestion of sewage sludge and organic waste has enhanced the biological activities in the digester at Zirl WWTP, just as has been observed from other case studies. Consequently, both sludge and biowaste are more efficiently degraded, and overall sludge production is lower (and overall biogas yield higher) than the sum of individual digestion of the same substances. Regarding polymer consumption for sludge dewatering, the results are inconclusive: over the years, specific polymer consumption has fluctuated within a range of 10–17 g polymer/kgDS. No clear attribution of higher or lower values to the impacts of co-digestion is possible.

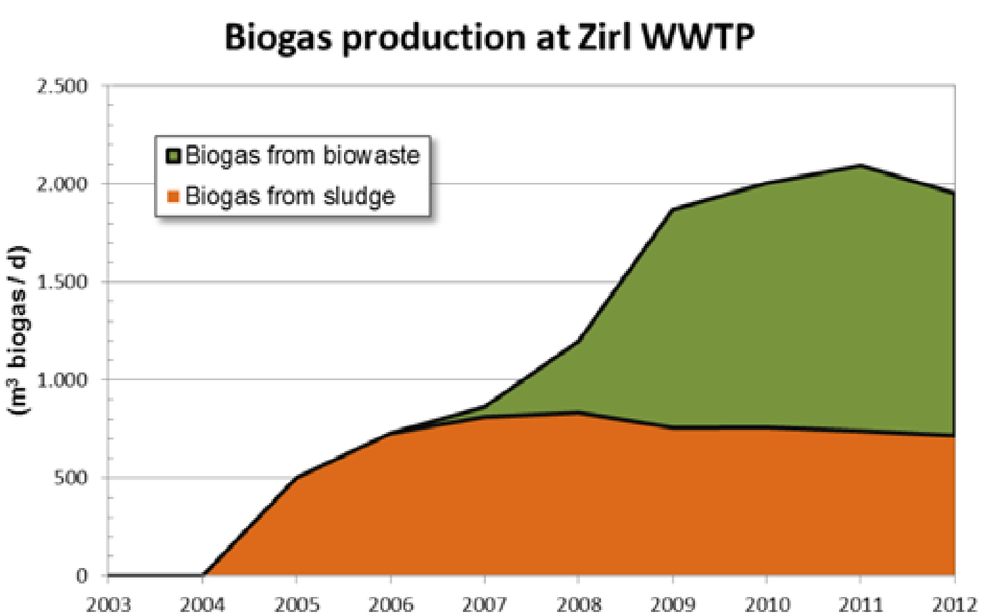
The two key loading parameters of the digester are the typical retention time including waste, which was >20 days; and the organic loading equaled about 2.5 kg VS/m^3/d. Both those values are thus within the range recommended in table CS1-4.

5.2.2. Biogas production and potential for energy generation

Biogas production

Figure CS5-7 shows biogas production in the digester at Zirl WWTP. Startup of the digester was during calendar year 2005. Hence, biogas production from sludge reached its full capacity without co-digestion only in 2006. In 2007 and 2008, only small quantities of organic waste were co-digested. Consequently, biogas production increased, but not too much by then. Since 2009, large solids loads of organic waste have been co-digested, and more biogas has been produced from this waste than from sludge ever since.

Figure CS5-7: Daily biogas production at Zirl WWTP

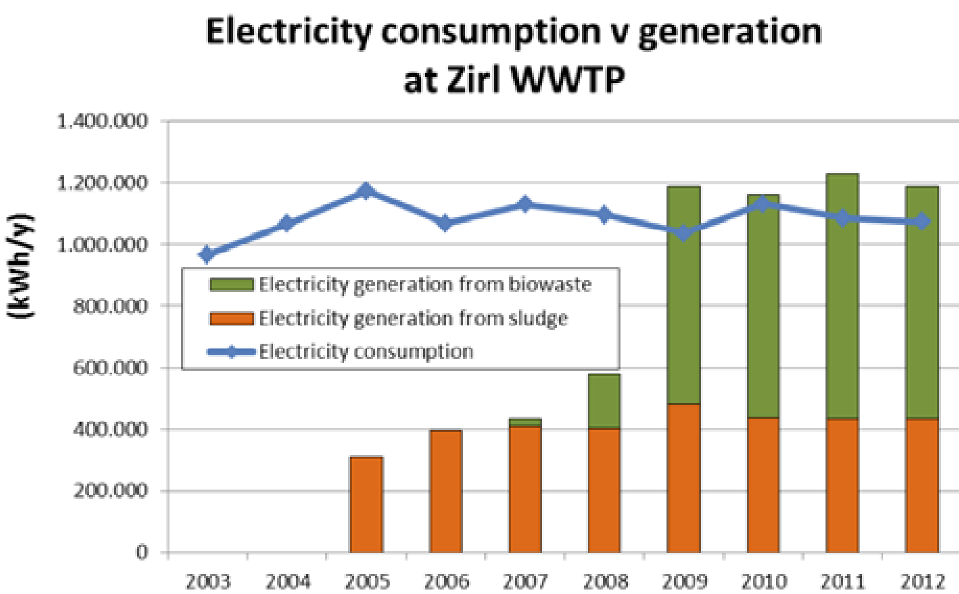


Source: Abwasserverband (AV) Zirl et al.

Potential for energy generation

The increased biogas quantities changed the potential for electricity generation drastically. Figure CS5-8 demonstrates this clearly.

Figure CS5-8: Power generation from biogas, compared to electricity consumption at Zirl WWTP



Source: Abwasserverband (AV) Zirl et al.

Initially, electricity generation from Zirl WWTP's sludge sources covered only about 40 percent of its electricity consumption. Yet ever since 2009—when co-digestion increased greatly—more electric power has been produced from biogas than the plant consumes. It is worthwhile mentioning that the operator is intentionally not increasing electricity production further, since feed-in tariffs for electricity supplied to the public grid are low in Austria, and financially unattractive. Thus, the operator's main target is full electricity coverage of the WWTP's needs.

When analyzing the biogas yield as a function of input VS, it turns out that biogas production is, in fact, higher than expected. Hence, a similar conclusion can be drawn as that previously drawn for sludge production: the mutual digestion of sewage sludge and organic waste enhances the biological activities in the digester; consequently, both sludge and biowaste are more efficiently degraded, and overall sludge production is lower (and overall biogas yield higher) than the sum of individual digestion of the same substances. A clear quantification of the effect for each individual input is not possible with the data available, but the overall trend is beyond doubt.

The specific electricity consumption at Zirl WWTP in recent years has equaled about 25 kWh/PE_{60}/y. This is a good result as compared to average results in Europe, according to case study 1, but it is in line with actual recommendations for this plant size and N elimination treatment target (compare annex 1, table A-1: consumption target = 30 kWh/PE_{60}/y; ideal = 23 kWh/PE_{60}/y).

The specific biogas production prior to co-digestion was 17.5 L biogas/PE_{60}/d. This value is also in line with typical values presented in figure CS1-2 and table CS1-2. With co-digestion, this value increased to about 45–50 L/PE_{60}/d.

Table CS5-4 presents a summary of key biogas and power generation parameters.

Table CS5-4: Biogas and power generation potential of co-digestion at Zirl WWTP

Biogas production and Power Generation	Retention time in PST (h) 0.5–1.0
Biogas production	
- N elimination (L/PE_{60}/d)	**17.5 (sludge) + 30 (waste)**
- C elimination (L/PE_{60}/d)	**—**
Electric efficiency CHP (%)	25 and 35
Thermal efficiency CHP (%)	50 and 55
Calorific value of biogas (kWh/m^3)	6.2
Power generation	
- N elimination (kWh/PE_{60}/year)	**11 (sludge) + 19 (waste)**
- C elimination (kWh/PE_{60}/year)	**—**
Thermal energy generation	
- N elimination (kWh/PE_{60}/year)	18 (sludge) + 32 (waste)
- C elimination (kWh/PE_{60}/year)	—
Total energy generation	
- N elimination (kWh/PE_{60}/year)	29 (sludge) + 51 (waste)
- C elimination (kWh/PE_{60}/year)	—

Source: Authors' calculation.
Note: L/PE_{60}/d x 16.67 = L/kg BOD_5/d; kWh/PE_{60}/y x 16.67 = kWh/kg BOD_5/y.

Zirl WWTP is not unique in its biogas production from co-digestion; other WWTPs report similar results. For instance, Schwarzenbeck and others (2008) report on a municipal WWTP that increased its electricity production with co-digestion to 113 percent, on average. And many of the references cited, together with tables CS5-7, and CS5-9, present further successful examples, albeit frequently with smaller quantities of biowaste involved and thus less pronounced increases of electricity coverage.

5.2.3. Operation capacity needs, biogas safety

All issues generally associated with biogas safety and operation capacity needs have already been discussed in case study 1. For details, see section 1.2.3.

No particular extra capacities or precautions are required with co-digestion of organic waste. The only exception is that, in pretreatment, any substances that may settle in the digester need to be properly eliminated and/or reduced in size so they will leave the digester together with the digested sludge.

5.2.4. Institutional aspects, energy costs

The institutional energy aspects relevant in Central Europe have already been discussed in case study 1 for Austria, where Zirl WWTP is located. For details, see section 1.2.4.

Some other relevant institutional aspects are the following:

- Collection and pretreatment of the organic waste are carried out by a private company.
- All investments required for waste collection and pretreatment were made by the private company.
- Zirl WWTP receives the sludge free of charge.

The advantage of this cooperation for the private company is that it avoids the high cost of waste disposal and has shorter transport distances. The mixing of sludge and organic feedstock requires a formal approval by the relevant authorities in Austria.

5.2.5. GHG reduction and CDM co-financing

The same principles apply as already described in case study 1. For details, see section 1.2.5.

5.2.6. CAPEX structure

CAPEX requirements for the co-digestion project at Zirl WWTP have been almost zero. Particularly since the pretreatment of the waste was to be carried out by a private company, not much remained to invest at the WWTP; the biowaste holding tank and its periphery installations were already in place. Therefore, only minor adjustments were made to the existing infrastructure, implying a CAPEX of less than US$6,800.

CAPEX requirements cannot be standardized for other plants, either. Very much depends on the specific organic waste, its specific pretreatment requirements, and the specific installations available that can be utilized. Still, the following thumb numbers can offer some orientation:

- *FOG installations:* A typical concept for FOG co-digestion would include the installation of a FOG holding tank that can be heated (in temperate climates only). This tank (for example, 30 m^3), complete with connection for delivery tanks, mixer in the tank, FOG grinding, feeding pump, and connecting pipe to digester, typically implies CAPEX on the order of US$68,000.
- *Reception and holding chamber:* Such a chamber is required when the organic waste is already pretreated and/or does not require pretreatment. In these cases, larger reception installations, into which trucks can dump their organic loads, may be required. These installations, complete with mixer, feeding, and so on, cost around US$135,000.
- *Mechanical pretreatment.* Some types of organic waste, such as municipal organic waste, do require pretreatment. CAPEX of a hammer mill, as utilized for the biowaste that is co-digested at Zirl WWTP,

depends on its capacity. A large installation with a capacity of about fifteen tons/hour (approximately 25 m^3/h), complete with all peripheral components, could cost about US$675,000.

5.2.7. OPEX structure

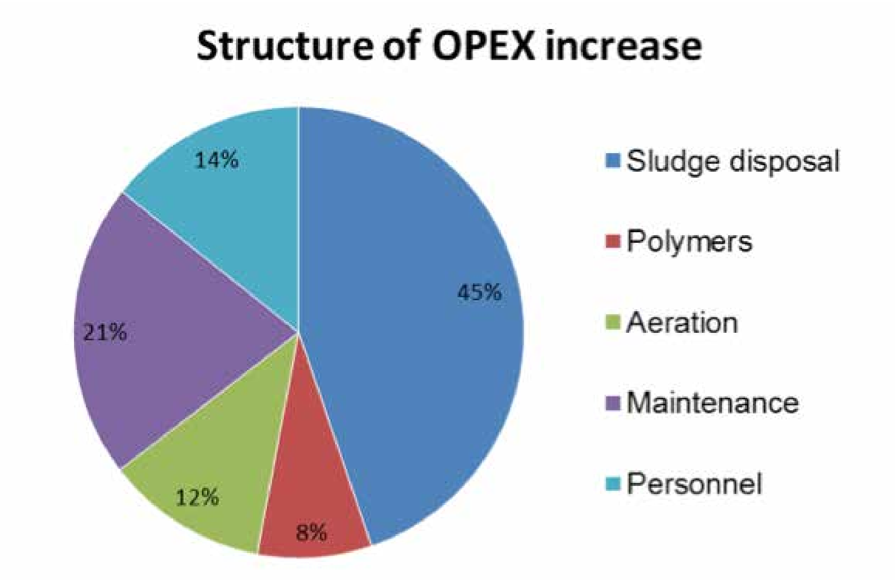
The OPEX of the co-digestion in Zirl WWTP has been analyzed in detail for the year 2012. In this analysis, the benefit of a reduced electricity bill was balanced against the extra OPEX of co-digestion. Details are presented in table CS5-5. Furthermore, the structure of "additional cost" items, which increase OPEX, is presented in figure CS5-9.

Table CS5-5: OPEX implications of co-digestion at Zirl WWTP in 2012

	OPEX (US$)
BENEFIT	
Reduced electricity bill	**-88,425**
ADDITIONAL COST	
Additional sludge disposal cost	31,725
Additional polymer required for sludge dewatering	5,805
Additional aeration requirement due to additional N return load	8,235
Additional maintenance (CHP, pumps, etc.)	14,985
Additional personnel cost	10,125
Subtotal	**70,875**
BALANCE	
TOTAL	**-17,550**

Source: Abwasserverband (AV) Zirl et al.

Figure CS5-9: Structure of OPEX increasing cost items of co-digestion at Zirl WWTP in 2012



Source: Abwasserverband (AV) Zirl et al.

While cost-saving effects result from a reduced electricity bill, the main cost-increasing effect is associated with the additional digested sludge (sludge disposal + polymers), which makes up about 55 percent of all OPEX increases.

Related to the annual quantities of 3,300 m^3 organic waste, the total extra OPEX equals US$21.5/$m^3$ for the organic waste added. The nearby Strass WWTP reported extra OPEX for co-digestion of US$30.4/$m^3$ (Dengg 2003). Thus, as a general rough estimate, additional overall OPEX of about US$27/$m^3$ organic waste added seems to reflect central European conditions well.

At Zirl WWTP, the benefits from electricity savings outweigh the additional OPEX and lead to net savings of US$17,550. Related to the annual quantities of 3,300 m^3 organic waste, the net savings equal about US$5.4/$m^3$ organic waste added. When comparing the overall financial benefits in electricity cost savings, at Zirl WWTP the actual net savings amount to 20 percent of the electricity savings. This seems quite a typical result for central European conditions. Svardal and Haider (2010), after analyzing five WWTPs with co-digestion in the same region, conclude this ratio may vary between 0 and 50 percent. Hence, specific local conditions have a significant financial impact, even in a narrowly defined region.

Tentatively, in regions where sludge disposal is not as expensive as at Zirl WWTP (unit disposal cost of US$85/ton) and where personnel is cheaper, the OPEX increases can be substantially less, and the overall cost balance could be considerably more attractive.

It remains to be mentioned that in this calculation, the disposal cost of organic waste that would be incurred if it were not co-digested has not been factored into the considerations, since Zirl WWTP receives the sludge free of charge from the private company. This could be quite different at other locations.

Likewise, not included are the additional thermal energy gains from the increased biogas production, since at Zirl WWTP it has no financial value.

5.2.8. Viability of investment in biogas utilization

In the case of Zirl WWTP, the application of co-digestion makes economic sense; it achieves an annual OPEX reduction of US$17,550 without additional investments.

Moreover, it reduces the dependence of the WWTP on outside power supply. Thus, even in the event of power blackouts or any other problems with the public power supply, the WWTP can continue operating at a normal level.

Table CS5-6 summarizes several cost indicators from case study 5.

Table CS5-6: Cost indicators for co-digestion at Zirl WWTP

		Case study 5
Average influent load	$PE_{60,avg}$	42,950
Average CAPEX	EUR/PE_{60}	0.1
Average OPEX reduction	EUR/PE_{60}/y	-0.3
Average CAPEX	US$/$PE_{60}$	0.1
Average OPEX reduction	US$/$PE_{60}$/y	-0.4

Source: Authors' calculation.

Note: 1 PE_{60} = 1.0 cap in Austria; US$/$PE_{60}$ x 16.67 = US$/kg BOD_5.

In sum, co-digestion at Zirl WWTP is *financially viable*, and it increases operation safety.

5.3. Conclusions for co-digestion of organic waste in EAP countries

What remains unchanged in EAP (as compared to case study 5)

Most likely, independent of location, after the introduction of co-digestion no changes will be required to the wastewater, the sludge, or the biogas train.

Wastewater treatment requirements matter only if enhanced nitrogen (N) removal is required. Via the filtrate from sludge dewatering, an extra load of N is recycled, which has to be removed from the wastewater train where nitrogen standards exist. This implies a need for additional treatment capacity. A case-specific assessment of its quantities and implications is always needed. But, as case study 5 has shown, nutrient removal WWTPs might be able to cope with this requirement without any expansion works even when feeding large quantities of extra feedstock.

Neither the widespread wastewater dilution in EAP nor the continued use of septic tanks has an impact on co-digestion.

Not much change is expected in the potential for increased biogas production when comparing European results with EAP results. After all, the digesters are closed systems with a controlled environment that works comparably well at any location worldwide. The additional waste will be partially destroyed, producing additional biogas. The extent of that increase depends on quantities and qualities of the extra feedstock fed into the digesters. Thus, the electricity generation potential will also increase accordingly, to very much the same extent.

CAPEX is usually required for pretreatment and feeding of the extra waste. The necessary extent and cost are highly case sensitive and can range from very low amounts to substantial investments.

OPEX savings are surprisingly low in case study 5. This result may be drastically different in EAP, foremost because the extra sludge disposal and manpower are much cheaper there.

The regulatory framework in EAP is still in development. For instance, Vietnam only recently introduced legislation defining feed-in tariffs for electricity generated from biomass and from landfill biogas; this legislation has also obliged the power utility to buy such feed-ins. Another ongoing activity in Vietnam is the drafting of technical guidelines for energy recovery from sewage sludge.

Success stories in EAP

Co-digestion is not a traditional approach in Asia or EAP countries. This is definitely connected to the fact that anaerobic digestion in general is not yet widespread there. But the interest in co-digestion is growing. For instance, Chen and others (2013) report from Singapore on co-digestion experiments with food waste. The biogas yield they found is on the upper end of the range indicated for this kind of feedstock in table CS5-10. This clearly shows that the potential for co-digestion exists in EAP.

In Vietnam particularly, co-digestion of fecal sludge with sewage sludge is considered of interest. The Hanoi University of Civil Engineering (HUCE) is conducting a research program on that topic at present (Viet Anh et al. 2014); final results are expected next year. A preliminary result showed a wide variety of fecal sludge qualities, with some being quite suitable for co-digestion. The co-digestion of fecal and sewage sludge also shows the same performance- enhancing effects highlighted in this technical note for other co-digestion feedstocks in comparison to separate digestion.

Such co-digestion of sewage sludge with fecal sludge could prove a promising combination for many locations in EAP, where large numbers of septic tanks will prevail for many years to come. Keeping in mind that fecal sludge quality does not only vary within the same city but also from region to region, more research is definitely needed.

Table CS5-7: Co-digestion of organic waste in anaerobic digesters: General design parameters and key characteristics of sludge management

DIGESTED SLUDGE PRODUCTION	The total digested sludge production from co-digestion can be calculated as the digested sum of its individual inputs. It is worth mentioning that this approach may lead to slight overestimates of total digested sludge production. It has been confirmed by various studies and applications (Dengg 2013; Kusowski et al. 2013; Iacovidou et al. 2012; Johnson et al. 2011; Svardal and Haider 2010; Callegari 2010; Zupancic et al. 2008; Felde et al. 2005; Jansen et al. 2004), that the mutual digestion of sewage sludge and organic waste enhances the biological activities in the digester. Consequently, both sludge and biowaste are more efficiently degraded, and overall sludge production is lower (and overall biogas yield higher) than the sum of individual digestion of the same substances. Simple design tools for quantification of these effects without practical experiments are not available. Therefore, the common design approach is to add up the following components: • Digested sludge production from waste sludge • See table CS1-5. • Digested sludge production from organic waste The theoretical sludge production from organic waste digestion can be estimated by assuming a typical percentage for VS destruction (see table CS5-10). An estimate for the digested biowaste DS can then be derived from total biowaste DS input to the digester, minus VS destroyed.
DIGESTED SLUDGE DEWATERING	The practical implications of co-digestion for sludge dewaterability do not show a uniform trend. Both improved dewatering results and reduced polymer consumption for conditioning and reduced dewatered DS and increased polymer consumption are reported. For design considerations it is deemed acceptable to assume no relevant changes of sludge dewatering characteristics (that is, to assume the same DS and specific polymer consumption) after the introduction of co-digestion, as compared to prior conditions.
FILTRATE QUALITY FROM SLUDGE DEWATERING	The filtrate quality from sludge dewatering is affected by co-digestion. Whereas the expected impacts on carbon parameters (BOD_5, COD, DS) and phosphorus are mostly considered negligible for municipal WWTPs, the additional nitrogen (N) that is released from the destroyed organic waste and returned to the wastewater train via the sludge dewatering filtrate may be worth considering.

If the WWTP in question is only designed for carbon removal, this additional N load is not crucial for compliance with the required effluent quality standards. However, due to the fast nitrification in warm climates, it will nonetheless increase energy consumption for aeration. If the WWTP is designed for enhanced N removal, the issue becomes even more important. In this case, the additional N load has to be duly considered in the design of the wastewater train.

The additional N load can be estimated by assuming all N contained in the destroyed VS fraction of the organic waste ends up in the filtrate. (This is a safe assumption, since a minor percentage of this "destroyed" N will also be used for biomass synthesis.) For design purposes, it is hence important to know the N content of organic waste and the typical VS destruction in co-digestion. The former is presented in table CS5-8, and the latter can be taken from table CS5-10. If of interest, the additional P load that is returned to the wastewater train from organic waste co-digestion may be calculated the same way.

Table CS5-8: N and P content of various types of organic waste

Organic waste	N_{total} (% of DS)	P_{total} (% of DS)
FOG from grease traps	**2** (1.5–3.7)	**0.3** (0.1–0.7)
Waste food from restaurants, canteens, etc.	**2** (0.6–5.0)	**0.7** (0.1–1.5)
Bakery waste	**2**	**0.7**
Whey from dairy	**1–2**	**—**
Agro-industrial waste (e.g., from sugar mills, starch mills, breweries, etc.)	**1–13**	**0.5–2.6**
Municipal organic waste	**1.5** (0.5–2.7)	**0.5** (0.3–0.8)

Sources: Based on Traversi et al. 2013; ARAconsult 2009; Huber et al. 2007; Stabnikova and Wang 2006; Felde et al. 2005; DWA 2003b; Billmaier et al. 2001.

Table CS5-9: Co-digestion of organic waste in anaerobic digesters: General design parameters and key characteristics of energy management

BIOGAS DESIGN

Biogas production in co-digestion originates from different sources, which can be assessed separately and added together to come up with total biogas production:

- *Biogas from waste sludge*

See table CS1-5.

- *Biogas from organic waste*

The biogas production from organic waste can be estimated using the three different approaches described below. The method of choice usually depends on data availability.

(i) Biogas yield from the digestion of kgVS

Table CS5-10: Biogas yield from the digestion of kgVS

Organic waste	Dry Solids DS (% of weight)	Volatile solids VS (% of DS)	Volatile solids destruction in co-digestion (% of VS)	Biogas yield (L / kg VS)	Biogas yield (L / kg $VS_{destroyed}$)	Methane content (%)
FOG from grease traps	**5** (2-30)	**90** (70-95)	**85** (70-90)	**1000** (600-1600)	**1100** (700-2000)	**75** (70-85)
Waste food from restaurants, canteens, etc	**10** (5-30)	**90** (70-96)	**80** (60-95)	**800** (500-1000)	**1000** (700-1250)	**60** (50-65)
Bakery waste	**75**	**97**	**90**	**570**	**640**	**53**
Whey from dairy	**5** (5-6)	**95** (90-95)	**95** (94-99)	**800** (740-950)	**850** (800-1000)	**53**
Agro-industrial waste (e.g. from sugar mills, starch mills, breweries, etc)	**10-90**	**60-95**	---	**400-800**	---	**55-70**
Municipal organic waste	**25** (10-40)	**60** (30-80)	**75**	**500** (300-650)	**650** (400-950)	**60** (55-70)

Sources:

FOG: *Miot et al. 2013; Norgaard et al. 2013; Schafer et al. 2013; VSA 2012b; Johnson et al. 2011; Felde et al. 2005; DWA 2003b; Loll 2001; Billmaier et al. 2001; Schmelz 2000.*

Waste food: *Chen et al. 2013; Schafer et al. 2013; VSA 2012b; Zupancic et al. 2008; Stabnikova and Wang 2006; Schmelz 2003; DWA 2003b; Loll 2001; Braun 2001.*

Bakery waste: *ARAconsult 2009; Huber et al. 2007.*

Whey: *Traversi et al. 2013; Baubüro and Syneco 2012.*

Agro-industrial waste: *DWA 2003b.*

Municipal organic waste: *VSA 2012b; Schmelz 2003; DWA 2003b; Loll 2001.*

(ii) Biogas yield from the digestion of kgCOD

The methane yield can be calculated with a specific value of 350 L CH_4/kg $COD_{destroyed}$ (Nowak and Ebner 2013; Urban and Scheer 2011; Svardal and Haider 2010; DWA 2002). This methane yield may then be converted into biogas yield by taking the methane content of biogas (see table CS5-10) into account. For $COD_{destroyed}$ as a percentage of COD_{total}, similar percentages as indicated for $VS_{destroyed}$ in table CS5-10 can be applied.

Of course, COD and VS of organic waste are interrelated. However, the ratio between these parameters is not a constant, since it depends on the material's composition. Typical ratios are, for instance, COD/VS ≈ 2.0–2.5 for FOG, and COD/VS ≈ 1.4–1.6 for waste food (Urban and Scheer 2011; Svardal and Haider 2010).

(iii) Biogas yield from the digestion of carbohydrate/proteins/lipids

Table CS5-11: Biogas yield from the digestion of carbohydrates, proteins, and lipids

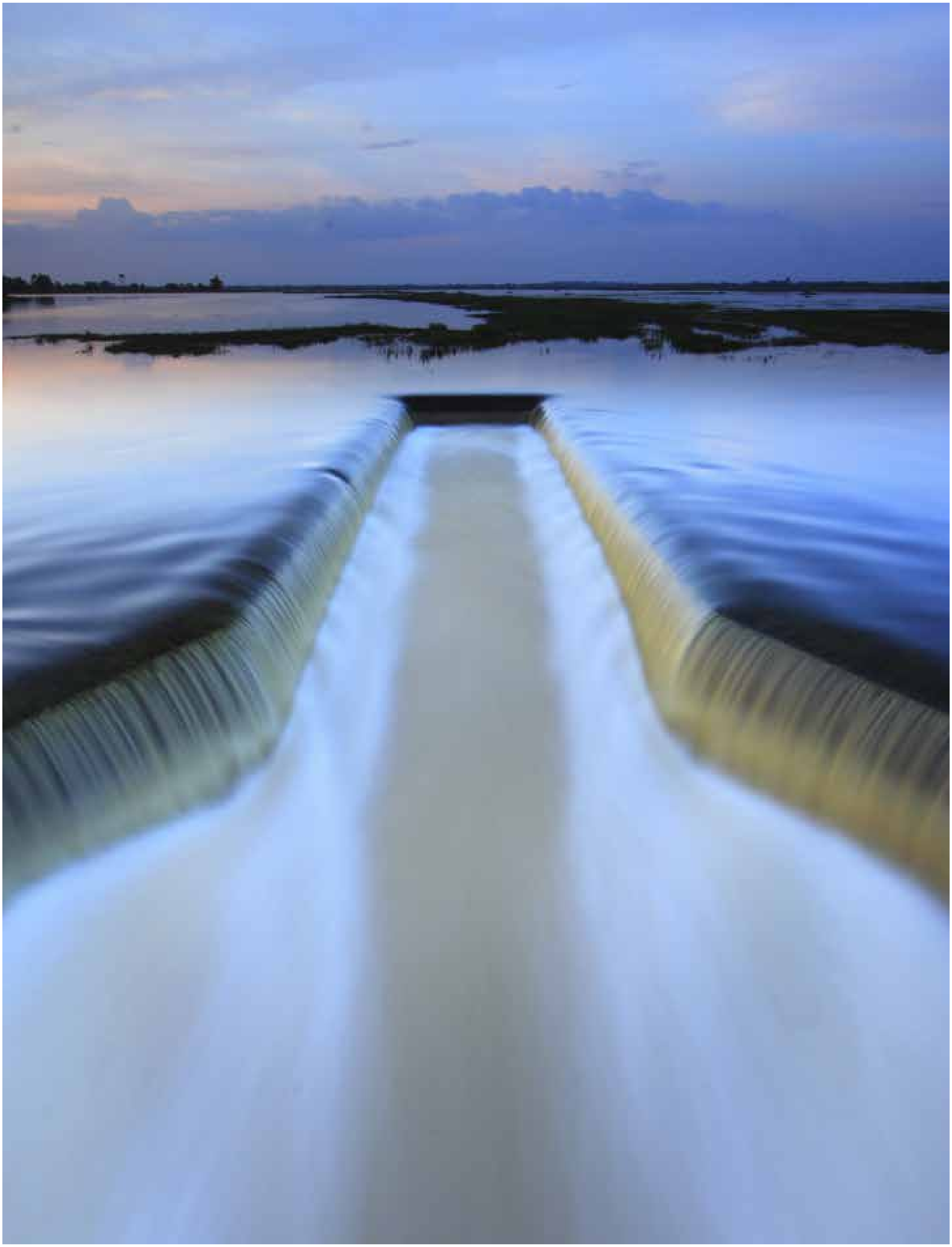
	Biogas yield (L/kg VS)	Methane content (%)
Carbohydrates	830	50
Proteins	720	71
Lipids	1,430	70

Source: DWA 2002.

The biogas yields in table CS5-11 represent the theoretical maximum yield, if all VS were completely digested. The numbers presented are taken from DWA (2002), but quite similar theoretical values are cited by Miot and others (2013) and Urban and Scheer (2011). The practical biogas yields, though, are lower, since (a) not all VS is fully destroyed (compare table CS5-10); (b) the chemical composition of the substrates may vary within a certain range; and (c) a fraction of the substrate is always used for cell synthesis. Hence, when calculating the biogas yield for a specific organic waste, these additional aspects should be applied to reduce the biogas values derived from the data provided in table CS5-11. In design practice, usually only the very first of these impacts (a) is indeed phased into the calculations.

The overall methane content from the digestion of a specific organic waste can be calculated as the weighted result according to the respective methane contents provided in table CS5-11.

BIOGAS TREATMENT *See table CS1-5.*
GAS HOLDER *See table CS1-5.*
FLARE *See table CS1-5.*
BIOGAS UTILIZATION *See table CS1-5.*



CASE STUDY 6: ULTRASOUND SLUDGE DISINTEGRATION

6.1. BACKGROUND, PROCESS DESCRIPTION

6.1.1. Data sources

Sludge disintegration can be used for various purposes: reduction of foaming problems, reduction of sludge quantities, or increase of biogas production. In this case study, the focus is exclusively on the last.

Several sludge disintegration technologies are available on the market. The most common are based on mechanical, thermal, or chemical-thermal processes. Generally, the thermal and chemical-thermal processes (for example, the Cambi process and Pondus process) are not deemed very appropriate for the EAP countries on which this report focuses because they involve high-tech installations and require highly skilled operators. The mechanical disintegration technologies include, among others, lysate centrifuges, agitator bead mills, and **ultrasound sludge treatment**. Among these, the last is the most widely used. In Europe there are now about 100 WWTPs using this technology, and in Asia it has gained ground, particularly in the south. Furthermore, it is a small installation that is very easy to operate. For these reasons, this system was selected for the present case study.

Case study 6 presents the experiences of eight municipal WWTPs with the application of ultrasound sludge disintegration to enhance biogas production in mesophilic sludge digesters. All the analyzed plants are equipped with ultrasound installations from VTA Technologie GmbH (www.vta.cc), which is among the leading providers worldwide of these installations. Similar products from other manufacturers of ultrasound equipment are also available on the market.

Since ultrasound sludge disintegration is implemented in closed reactors, treating conventional waste activated sludge as can be found anywhere in the world, the results from this case study can be transferred to any other location worldwide.

6.1.2. Wastewater management

Wastewater treatment at all the WWTPs analyzed in this case study is done by CAS. Since sludge disintegration is applied in the sludge treatment train, there are no changes to the wastewater train. Only one aspect may require attention: when using ultrasound sludge disintegration, the nitrogen (N) load in the filtrate from sludge dewatering increases. Those plants that require enhanced N removal will hence have to deal with this additional N load in one way or another. Yet, since the additional loads are relatively small, the impacts are small as well.

6.1.3. Sludge management

All the WWTPs of this case study employ CAS technologies, combined with primary sedimentation. Therefore, all these plants produce two types of sludge: primary sludge and waste activated sludge (WAS).

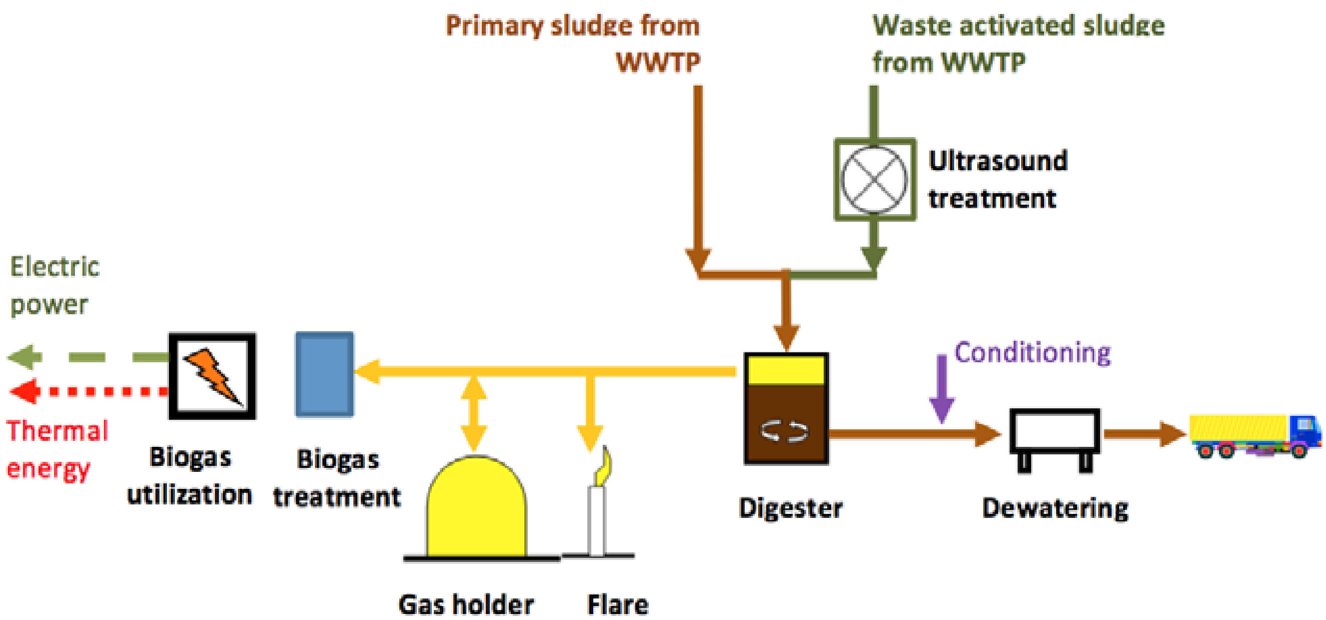
Ultrasound sludge disintegration (USD) for biogas enhancement is based on the principle that microbial cell walls are destroyed. The (easily digestible) cell liquors are released and contribute to additional biogas production. Hence, ultrasound disintegration

[4] The disintegration (cell wall destruction) effect of ultrasound treatment is based on cavitation. Ultrasound oscillator elements are usually positioned inside a small disintegration reactor. The emitted ultrasonic waves cause a periodic compression and decompression in the sludge matrix. Locally this causes extreme conditions with high pressures and temperatures. Cavitation bubbles are formed and implode again. Depending on the energy applied, the extent of cell lysis of the sludge's microorganisms can be controlled.

is exclusively applied to waste activated sludge, where microorganisms dominate the sludge mixture; treating primary sludge would not increase the biogas yield, since the bulk of it is organic matter and not microorganisms.

Typically, USD is located after thickening of WAS and prior to digester feeding. The treated sludge is subsequently fed into the digester. Figure CS6-1 depicts the typical positioning of USD in sludge treatment for biogas enhancement.

Figure CS6-1: Simplified flow scheme of ultrasound sludge disintegration for biogas enhancement



Source: Authors.

Specific characteristics of the USD system analyzed in this case study are the following:

- Frequency used: 25 kHz
- Typical retention time in the disintegration reactor ranging from 60 to 180 minutes
- Upstream flow of sludge inside the disintegration reactor
- DS content of 3–8 percent considered ideal for optimum efficiency

Key characteristics of the WWTPs and their USD systems analyzed in this case study are summarized in table CS6-1.

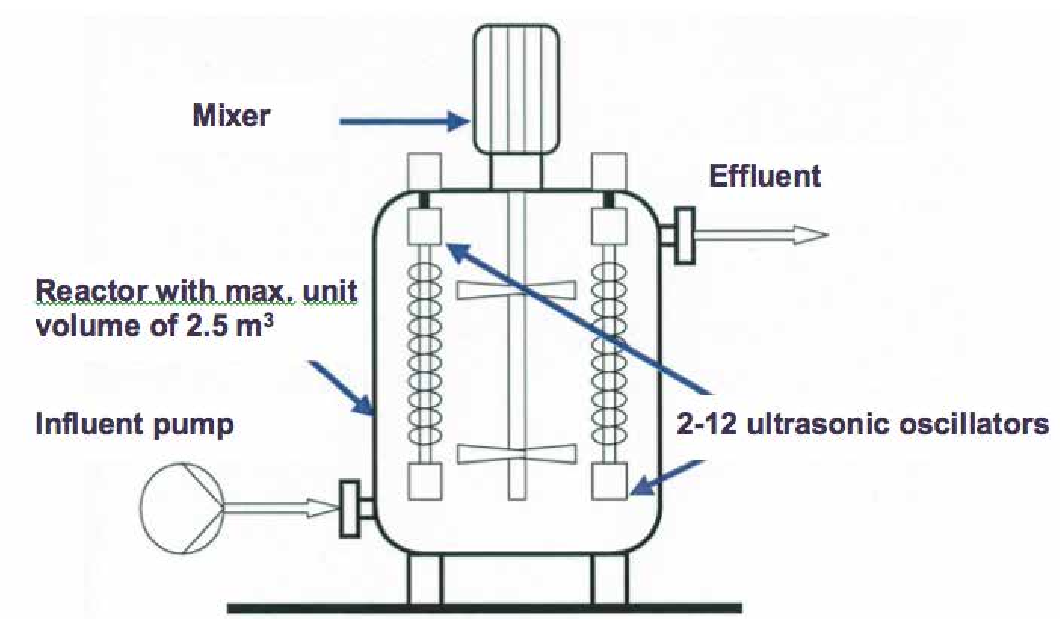
Table CS6-1: Key characteristics of WWTPs with ultrasound sludge disintegration investigated for case study 6

	WWTP	Country	Capacity (PE_{60})	Sludge disintegrated (% of WAS)	Analysis period
1	Villach WWTP	Austria	200,000	30	1 year
2	Großostheim WWTP	Germany	35,000	30–100	1 year
3	Halle Nord WWTP	Germany	300,000	40–60	3 years
4	Miltenberg WWTP	Germany	95,000	30	2 years
5	Roth WWTP	Germany	65,000	80–100	0.5 years
6	Wasserfeld WWTP	Italy	40,000	40–60	1 year
7	Estavayer-le-Lac WWTP	Switzerland	110,000	70–100	0.2 years
8	Moossee-Urtenenbach WWTP	Switzerland	40,000	50–100	1 year
	AVERAGE		110,625	ca. 60 percent of WAS	ca. 1.2 years

Source: VTA—private communication 2014.
Note: 1 cap = 60 g BOD_5/d in Europe.

Figures CS6-2, CS6-3, and CS6-4 present, respectively, a schematic of the disintegration reactor, some photographs of implemented installations, and microscopic pictures of sludge before and after USD.

Figure CS6-2: Schematic of an ultrasound sludge disintegration reactor

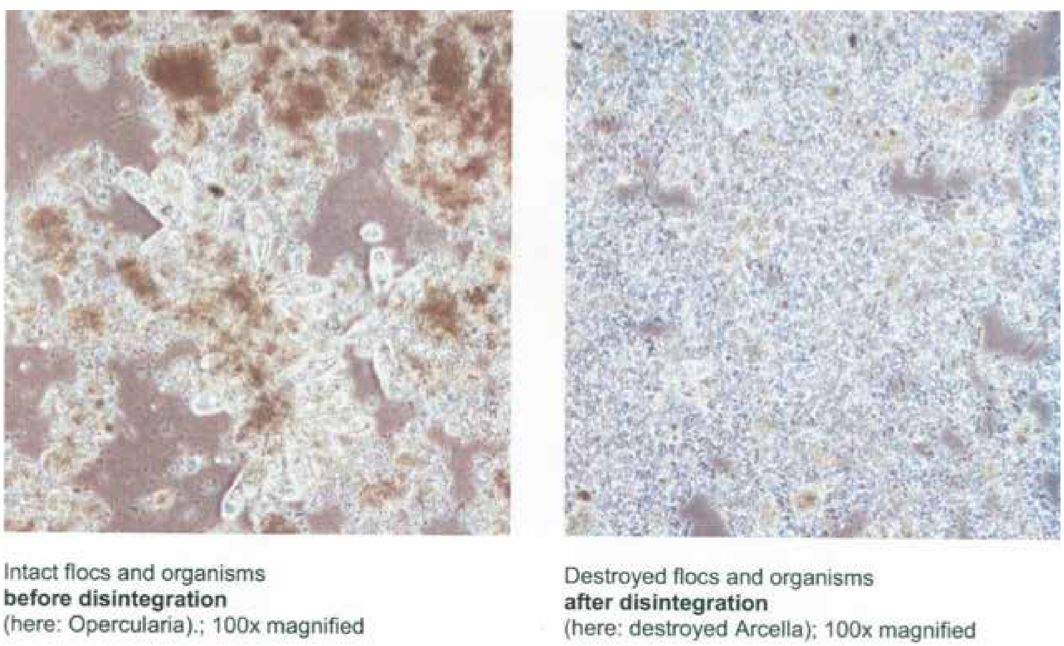


Source: Eder 2007.

Figure CS6-3: Implemented ultrasound sludge disintegration installations

Sources: VTA 2012; Eder 2007.

Figure CS6-4: Microscopic analysis of ultrasound sludge disintegration



Source: VTA 2012.

The design of anaerobic sludge digesters remains unchanged when introducing USD. Hence, for details on general design parameters and key characteristics of anaerobic digesters, see table CS1-4.

Relevant aspects prevailing for USD installations are summarized in table CS6-8.

6.1.4. Energy management

Two energy aspects have to be traded off when analyzing USD:

- Increased biogas, and thus increased energy production from biogas
- Additionally required energy input for the operation of USD

The crucial question is overall energy balance and the financial benefits compared to the CAPEX requirements, if it is positive; section 6.2 will analyze these issues.

For details on general design parameters and key characteristics of biogas systems combined with anaerobic sludge digesters, see table CS1-5. Typically, no changes are required in the biogas train when USD is installed. The increase in biogas is usually not so large that expanded capacities are required.

Specific, energy-related aspects of USD installations are summarized in table CS6-9.

6.2. Analysis

6.2.1. Wastewater influent, effluent, and other parameters of interest

Wastewater

No specific information regarding the wastewater characteristics of the WWTPs investigated for this case study is available. However, one can assume rather similar values to those in table CS1-1, which presented typical wastewater characteristics from the same region where the WWTPs of case study 6 are located.

Sludge

USD focuses on improved sludge treatment. Table CS6-2 summarizes the observed changes in sludge characteristics at the WWTPs of this case study. These results relate to an average observation period of 1.2 years, as summarized in table CS6-1.

From the results of this case study, the following can be observed:

Table CS6-2: Impact of ultrasound sludge disintegration on sludge characteristics

	WWTP	Observed changes					
		VS destruction in digester	VS content in digested sludge	Polymer consumption for dewatering	DS content in dewatered sludge	Sludge production (m^3/y)	Additional benefits
1	Villach WWTP	increase from 49% to 53%	decrease from 57% to 53%	decrease by 10%	no change	decrease by 5-6%	---
2	Großostheim WWTP	increase from 44% to 52%	decrease from 48% to 44%	no change	no change	decrease by 18%	Elimination of digester foaming.
3	Halle Nord WWTP	increase from 50% to 55%	decrease from 55% to 51%	no change	increase from 24.5% to 25.5 %DS	decrease by 5-6%	Elimination of digester foaming.
4	Miltenberg WWTP	increase from 47% to 57%	decrease from 53% to 50%	decrease by 10%	no change	decrease by 12%	---
5	Roth WWTP	increase from 48% to 55%	decrease from 56% to 50%	no change	no change	decrease by 5%	Stable digestion.
6	Wasserfeld WWTP	increase from 52% to 58%	decrease from 67% to 60%	decrease by 21.8%	increase from 23.1% to 26 %DS	decrease by 22%	---
7	Estavayer-le-Lac WWTP	increase from 51% to 58%	decrease from 66% to 58%	no change	no change	decrease by 12%	Elimination of digester foaming.
8	Moossee-Urtenenbach WWTP	increase from 50% to 56%	decrease from 53% to 50%	no change	increase from 26 % to 27 %DS	decrease by 15%	---

Source: VTA—private communication 2014.

- *Enhanced VS destruction during digestion:* VS destruction is clearly enhanced. VS destruction increases on average from 49 percent to 56 percent. This brings about a VS content in the digested sludge that is about five (three to eight) absolute percentage points lower than before USD. Comparable results are also cited in the literature (Schmelz and Müller 2004).
- *Improved DS in dewatered sludge:* Only three WWTPs report improved dewatering characteristics: two cite an increase of absolute dewatered DS of 1 percent, and one cites an increase of almost 3 percent. In all five other cases, dewatered DS is reported unchanged. Combining the observed reductions of dewatered sludge quantities (see below) with the observed VS destruction (see above) hints at an improvement of dewatered DS by an absolute +1 percent DS. Hence, all in all, an assumption of 0–1 percent DS increase appears realistic.
- *Reduced polymer conditioning in sludge dewatering:* While some WWTPs report reduced polymer consumption, others do not notice any changes. Some studies have even found increased polymer consumption; for example, Schmelz and Müller (2004) report an increase by 10 percent, so there is not a clear trend. For the time being, then, it is recommendable to assume no changes of polymer consumption after the introduction of USD.
- *Dewatered sludge production:* There is a general trend toward reduced sludge quantities of 12 percent on average (range: 5–22 percent).
- *Increased nitrogen (N) return load in filtrate from sludge dewatering:* This aspect went mostly unmentioned in the reports on the WWTPs of this case study. It is hypothesized that this is due to the combined effects of (a) little attention paid to this return load and (b) no significant changes in return load. Therefore, an increase of N return load by 5 percent, as found by Schmelz and Müller (2004), may be a realistic estimate.

6.2.2. Biogas production and potential for energy generation

Biogas production

Table CS6-3 and Figure CS6-5 show that USD treatment increased the biogas production of the eight WWTPs of this case study by 24 percent on average (range: 12–33 percent). This increase equaled about 100 (50–140) L/kgVS_{added}.

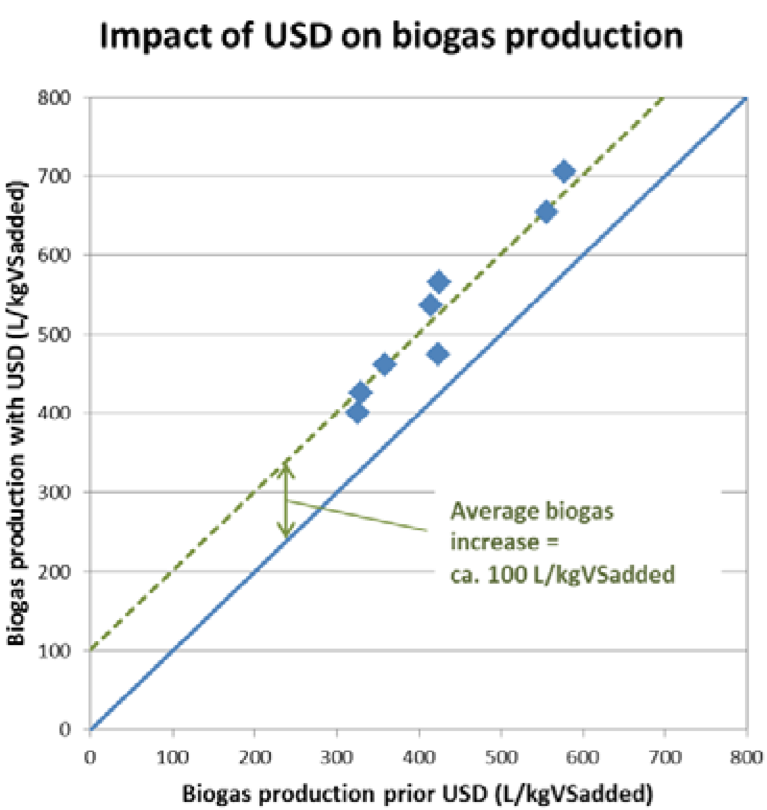
These results relate to an average observation period of 1.2 years, as summarized in table CS6-1.

Table CS6-3: Impact of ultrasound sludge disintegration on biogas production

	WWTP	Biogas production without USD	Biogas production with USD	Biogas increase	
		(L/kgVS$_{added}$)	(L/kgVS$_{added}$)	(L/kgVS$_{added}$)	(%)
1	Villach WWTP	556	654	98	18
2	Großostheim WWTP	425	565	140	33
3	Halle Nord WWTP	423	475	52	12
4	Miltenberg WWTP	329	425	96	29
5	Roth WWTP	414	537	123	30
6	Wasserfeld WWTP	358	461	103	29
7	Estavayer-le-Lac WWTP	577	705	128	22
8	Moossee-Urtenenbach WWTP	326	400	74	23
	AVERAGE	**426**	**528**	**102**	**24**

Sources: Weimer et al. 2005; Ullrich and Eder 2006; Wernitznig and Eder 2006; Petril 2006; Rausch 2007; VTA 2010; Casazza 2010; Resch 2011; VTA—private communication 2014.

Figure CS6-5: Impact of ultrasound sludge disintegration on biogas production



Sources: Weimer et al. 2005; Ullrich and Eder 2006; Wernitznig and Eder 2006; Petril 2006; Rausch 2007; VTA 2010; Casazza 2010; Resch 2011; VTA—private communication 2014.

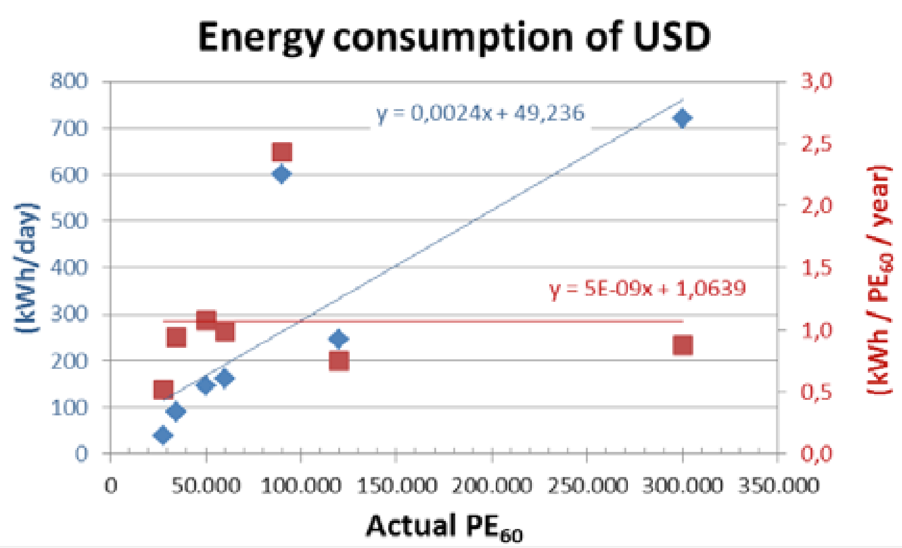
Potential for energy generation

Since the biogas characteristics do not change with USD, the biogas increase by 24 percent (12–33 percent) can be directly translated into additional energy generation of **24 percent (12–33 percent)**.

Energy consumption

The additional power generation is partly consumed for the power requirements of USD installations, as depicted in figure CS6-6 and table CS6-4.

Figure CS6-6: Energy consumption of ultrasound sludge disintegration



Sources: Weimer et al. 2005; Ullrich and Eder 2006; Wernitznig and Eder 2006; Petril 2006; Rausch 2007; VTA 2010; Casazza 2010; Resch 2011; VTA—private communication 2014.

Table CS6-4: Energy consumption of ultrasound sludge disintegration

		Actual load	Energy consumption for USD	
		(PE_{60})	(kWh/d)	(kWh/PE_{60}/y)
1	Villach WWTP	120,000	245	0.75
2	Großostheim WWTP	35,000	90	0.94
3	Halle Nord WWTP	300,000	720	0.88
4	Miltenberg WWTP	60,000	163	0.99
5	Roth WWTP	50,000	147	1.07
6	Wasserfeld WWTP	28,000	40	0.52
7	Estavayer-le-Lac WWTP	90,000	600	2.43
8	Moossee-Urtenenbach WWTP	35,000	90	0.94
	AVERAGE	**89,750**	**262**	**1.06**

Sources: Weimer et al. 2005; Ullrich and Eder 2006; Wernitznig and Eder 2006; Petril 2006; Rausch 2007; VTA 2010; Casazza 2010; Resch 2011; VTA—private communication 2014.

On average, USD's energy consumption amounts to 1.06 kWh/PE_{60}/year.

Given that biogas production increased by about 24 percent (12–33 percent), this enables an additional power production of 24 percent (12–33 percent). The typical electric power generation of CAS plants with mesophilic digesters is about 15–18 kWh/PE_{60}/y with N elimination (see table CS1-2). The increase of electricity generation thus equals about 4 (2–6) kWh/PE_{60}/y. It is possible to conclude that USD consumes roughly one-third of the electricity generated from the additional biogas. About two-thirds of the electricity increase remain as net benefit.

6.2.3. Operation capacity needs, biogas safety

All issues generally associated with biogas safety and operation capacity needs have already been discussed in case study 1. For details, see section 1.2.3.

No particular extra capacities or precautions are required for the O&M of the ultrasound installations. None of the WWTPs of this case study reported any particular problems. The USD automatically operates twenty-four hours a day, all year round. Operators usually just do routine inspections of pumping, mixing, and oscillator operation. Maintenance is limited to conventional pump and mixer maintenance and replacement of used oscillators after a couple of years.

It is worth mentioning that USD can help eliminate digester foaming.

6.2.4. Institutional aspects, energy costs

The institutional aspects prevailing in Central Europe have already been discussed in case study 1 for Germany and Austria. For details, see section 1.2.4.

The situation in Italy and Switzerland, where some of this case study's WWTPs are located, is very similar to that in Germany and Austria. Country-specific details of the actual regulations on renewable energy generation are not presented here, but can be found online at http://www.res-legal.eu/.

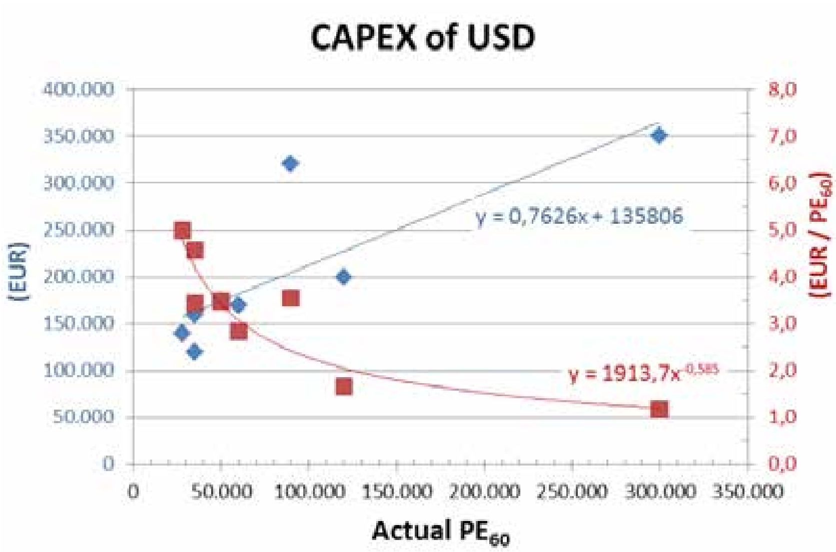
6.2.5. GHG reduction and CDM co-financing

The same principles apply as already described in case study 1. For details, see section 1.2.5.

6.2.6. CAPEX structure

CAPEX required for USD depends on plant size and shows a clear effect of economies of scale. Figure CS6-7 provides the USD cost numbers for all WWTPs of this case study. CAPEX for all investments required for a complete USD facility, including freight, installation, startup, engineering, and taxes, is on the order of EUR 3 (1–5)/PE_{60} (US$4 [1.4–6.8]/$PE_{60}$).

Figure CS6-7: CAPEX of ultrasound sludge disintegration



Sources: Weimer et al. 2005; Ullrich and Eder 2006; Wernitznig and Eder 2006; Petril 2006; Rausch 2007; VTA 2010; Casazza 2010; Resch 2011; VTA—private communication 2014.

6.2.7. OPEX structure

OPEX of USD is characterized by two opposing trends: some implications increase OPEX, and some decrease it. Table CS6-5 summarizes both effects.

Table CS6-5: Impact of ultrasound sludge disintegration on OPEX

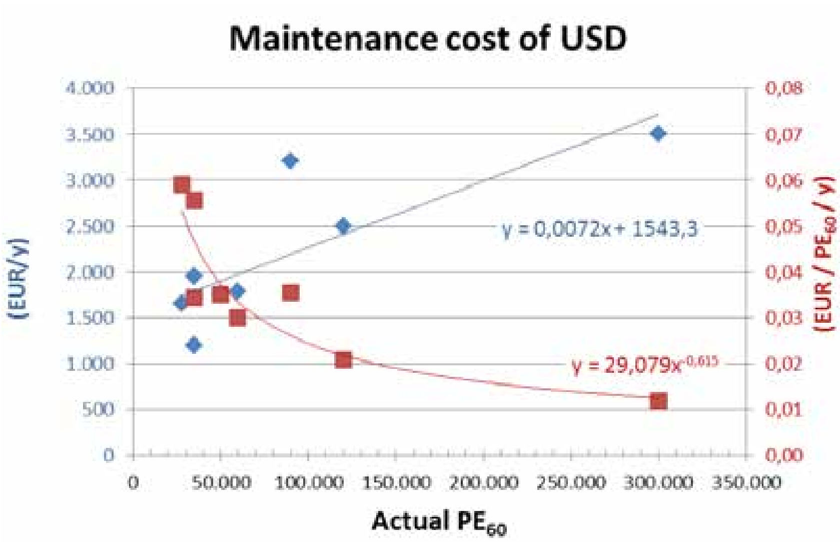
Item	OPEX decrease	OPEX increase
Electric power	Additional generation of about 24 percent (12–33 percent) in electric power, as compared to conventional CAS + digestion	Electric power required to operate USD: typically about one-third of the additionally generated power
Maintenance	—	Maintenance cost for new USD installations
Manpower	—	Manpower required for O&M of USD installation: typically 50 hours per year (Wernitznig 2010)
Sludge reuse/disposal cost	Reduction of sludge quantities by about 12 percent (5–22 percent) due to two combined effects: (a) enhanced VS destruction during digestion; (b) improved DS in dewatered sludge	—

Source: Authors' summary.

While electricity cost, manpower cost, and sludge reuse/disposal cost can be easily assessed with local unit cost, maintenance can be quantified according to figure CS6-8.

Under central European conditions, the dominant impact on OPEX comes from electricity. Taking all aspects into account, OPEX reduction found at the eight WWTPs of this case study turned out to be about EUR1.0 (0.3–1.7)/PE_{60}/y (US$1.4 [0.4–2.3]/$PE_{60}$/y), depending on local conditions.

Figure CS6-8: Maintenance cost of ultrasound sludge disintegration



Source: VTA 2014.

6.2.8. Viability of investment in biogas utilization

The amortization periods for investment into ultrasound sludge disintegration (USD) have been calculated by the various operators of the WWTPs in this case study. The outcomes are summarized in table CS6-6. All the results have been derived from NPV calculations, considering total life cycle cost.

Table CS6-6: Amortization periods reported for ultrasound sludge disintegration

	WWTP	Payback period (years)
1	Villach WWTP	≈4
2	Großostheim WWTP	≈5
3	Halle Nord WWTP	≈5
4	Miltenberg WWTP	≈4
5	Roth WWTP	≈5
6	Wasserfeld WWTP	≈6
7	Estavayer-le-Lac WWTP	≈4
8	Moossee-Urtenenbach WWTP	≈3
	AVERAGE	**5**

Sources: Weimer et al. 2005; Ullrich and Eder 2006; Wernitznig and Eder 2006; Petril 2006; Rausch 2007; VTA 2010; Casazza 2010; Resch 2011; VTA—private communication 2014.

Table CS6-7 summarizes several cost indicators from case study 6.

Table CS6-7: Cost indicators for ultrasound sludge disintegration at the WWTPs of case study 6

		Case study 6
Average influent load	$PE_{60,avg}$	89,750
Average CAPEX	EUR/PE_{60}	3 (1–5)
Average OPEX reduction	$EUR/PE_{60}/y$	1.0 (0.3–1.7)
Average CAPEX	$US\$/PE_{60}$	4 (1.4–6.8)
Average OPEX reduction	$US\$/PE_{60}/y$	1.4 (0.4–2.3)

Source: Authors' calculation.

Note: 1 PE_{60} = 1.0 cap in Central Europe; $US\$/PE_{60}$ x 16.67 = US\$/kg BOD_5.

At all eight WWTPs analyzed for this case study, it was concluded that the ultrasound sludge disintegration project is *financially viable*. Life cycle cost within the lifespan of new installations for USD (ten to fifteen years) was considered lower with the new installations than without. Average amortization is expected within about five years under European conditions.

6.3. Conclusions for ultrasound sludge disintegration in EAP countries

Wastewater treatment requirements matter only if enhanced nitrogen (N) removal is required. Via the filtrate from sludge dewatering, an extra load of N is recycled. A case-specific assessment is always needed of its quantities and implications. But, as case study 6 has shown, these implications are usually minor.

Neither the widespread wastewater dilution in EAP nor the continued use of septic tanks has an impact on USD. After all, USD and digesters are closed systems with a controlled environment that works comparably well at any location worldwide.

CAPEX is required for USD installations and should be similar in EAP and Europe.

OPEX savings are, in general, dominated by increased electricity generation and reduced sludge quantities. Since both these items feature lower unit cost in EAP than in Europe, absolute OPEX reductions should be lower in EAP.

In terms of financial viability, that means amortization periods may be somewhat longer in EAP than the typical five years found in Europe.

Success stories in EAP

Applications of USD are mostly limited to Europe to date.

In Shanghai, China, pilot tests have been done at Bailongang WWTP with ultrasound sludge disintegration of sludge from chemically enhanced primary treatment (CEPT). Not surprisingly, these results were not all that successful, since USD only works well with biological waste activated sludge (WAS; VA Tech Wabag 2006).

Recently, USD got a foothold in South Korea, where in 2011 a total of six large-scale USD facilities were successfully commissioned. These plants are called Daegu Bukbu WWTP, Daegu Seobu WWTP, Daegu Sincheon WWTP, Daegu Dalseo WWTP, Gangneung WWTP, and Gimhae WWTP. No operation data are known to the authors of this technical note.

ULTRASOUND SLUDGE DISINTEGRATION: TECHNICAL SUMMARY

Table CS6-8: Ultrasound sludge disintegration prior to anaerobic digesters for biogas enhancement: General design parameters and key characteristics of sludge management

USD CONSTRUCTION DETAILS	• Generally, ultrasound frequencies range from 20 kHz to 10 GHz. For sludge disintegration, 20–40 kHz is commonly used. • The typical lifespan of ultrasound oscillators = 45,000 operation hours. When running for twenty-four hours a day, that is equivalent to more than five years. The guaranteed lifespan usually is about 25,000 operation hours. • Modular systems, which can be easily extended • Low maintenance needs • Automated operation • Area requirement for disintegration reactor + pumps + control switchboard equal to about 5 m^2/100,000 PE_{60} • Minimum required room height = 2.8 m • Noise emissions: max 65 dB
DIGESTED SLUDGE PRODUCTION	*General:* The VS content in the digested sludge can be reduced by up to 25 percent (VTA 2012). Considering VS makes up about 50–60 percent of digested sludge, that means overall sludge quantities may be reduced by up to 10–15 percent. *Case study 6:* About 15 percent (10–20 percent) more VS was destroyed in the digesters with USD than without USD. Therefore, in absolute terms, the percentage of $VS_{destroyed}$ increased from 49 percent to 56 percent, on average. Furthermore, a reduction of dewatered sludge quantities of about 12 percent (5–22 percent) was observed. Since this reduction cannot be explained by the stronger VS destruction alone, it hints at an improvement of DS in dewatered sludge by an absolute +1 percent DS.
DIGESTED SLUDGE DEWATERING	*General:* A reduced VS content of the digested sludge should facilitate dewatering. Hence, the DS of dewatered sludge may increase, and dewatered sludge quantities (m^3) to dispose of/reuse may be reduced. The combined effect of reduced VS content (see above) and of improved dewatered DS is said to enable an overall sludge reduction of up to 20 percent (VTA 2012).

Additionally, polymer conditioning for dewatering may be reduced by up to 20 percent (VTA 2012).

Case study 6: The operators mostly did not report dewatered DS changes. Also, polymer consumption for sludge conditioning was usually not affected. Nonetheless, the reduction of dewatered sludge quantities hints at an increase of dewatered DS by an absolute +1 percent DS (see above item). This may not have been noticed by the operators, since it is a relatively minor impact that is not too apparent.

FILTRATE QUALITY FROM SLUDGE DEWATERING

General: The filtrate quality from sludge dewatering is affected by USD. Whereas the expected impact on carbon parameters (BOD_5, COD, DS) and phosphorus are mostly considered negligible for municipal WWTPs, the additional nitrogen (N) that is released together with the cell liquors and is returned to the wastewater train via the sludge dewatering filtrate may be worth considering.

If the WWTP in question is only designed for carbon removal, this additional N load is not crucial for compliance with the required effluent quality standards. Nonetheless, due to the fast nitrification in warm climates, it will increase energy consumption for aeration. If the WWTP is designed for enhanced N removal, the issue becomes even more important. In this case, the additional N load has to be duly taken into account in the design of the wastewater train.

The additional N load caused by USD can be estimated by assuming 100 g of additional N per additional $kgVS_{destroyed}$ (DWA 2009). Schmelz and Müller (2004) state that this leads to an increase of N return load by 5 percent.

Case study 6: No quantifiable impacts on increased return loads were observed at the WWTPs of this case study. This could be seen as a confirmation of the cited minor N increase of only 5 percent.

Table CS6-9: Ultrasound sludge disintegration prior to anaerobic digesters for biogas enhancement: General design parameters and key characteristics of energy management

ENERGY BENEFITS FROM USD	*General:* The energy benefits are directly proportional to the extra biogas generation. *Extra biogas:* Typical increases in biogas quantities are up to 30 percent (VTA 2012). For the electricity generation from that extra biogas, apply the principles of table CS1-5. Case study 6: USD treatment increased biogas production of the eight WWTPs of this case study by 24 percent (range: 12–33 percent). This equalled an increase by about 100 (50–140) L/kgVS$_{added}$. Given the typical electric power generation of CAS plants with mesophilic digesters of about 15–18 kWh/PE$_{60}$/y with N elimination (see table CS1-2), this equals an increase in electricity generation by 4 (2–6) kWh/PE60/y.
ENERGY REQUIREMENTS FOR USD	*General:* The following electricity consumption is caused by USD: *USD:* Typical electricity requirements of ultrasound sludge disintegration vary between 0.5 and 20 kWh/m^3 sludge with 5 percent DS treated. This is equivalent to about 0.01–0.4 kWh/kgDS with an average DS = 5 percent. These values hold true if 100 percent of the sludge is, indeed, treated (Müller 2010; DWA 2009). • *Peripheral installations (mixer, pumps):* Typically, 0.9 kWh/m^3 sludge is treated (DWA 2009). • *Aeration of N return load:* Additional aeration requirement due to N return load through filtrate = 1.95 kWh/kg N$_{eliminated}$ for fine bubble aeration (Müller 2010; DWA 2009). *Case study 6:* The overall energy consumption for all USD-related installations was found to be about 1.1 (0.5–2.4) kWh/actual PE$_{60}$/y. This is equivalent to approximately 30 percent of the additional electricity generated from the increased biogas quantities (see above).
	BIOGAS TREATMENT *See table CS1-5.* **GAS HOLDER** *See table CS1-5.* **FLARE** *See table CS1-5.* **BIOGAS UTILIZATION** *See table CS1-5.*