Part IV

Wastewater Treatment

Author: Gabor Doka, Doka Life Cycle Assessments

Doka G. (2007) Life Cycle Inventories of Waste Treatment Services. ecoinvent report No. 13, Swiss Centre for Life Cycle Inventories, Dübendorf, December 2007.

Final report of the project of a National Life Cycle Inventory Database "ecoinvent 2000" commissioned by BUWAL/BFE/ASTRA/BLW

Acknowledgements

I'd like to express thanks and gratitude towards the following people:

Max Maurer, EAWAG Dübendorf, for expertise on elemental flows in wastewater treatment plants and sludge disposal.

Jörg Feldmann, Department of Chemistry, University of Aberdeen, Scotland UK, for information on volatile (organo)metal emissions from wastewater treatment plants.

Alex Labhardt and Fränzi Huber, the authors of the original wastewater treatment inventory model published in (Zimmermann et al. 1996), which was in many parts the foundation of this work.

Summary

The goal of this study is to create life cycle inventories of the treatment of different wastewater compositions in Swiss municipal wastewater treatment plants WWTP. The system boundary includes the canalisation, the WWTP process itself and the disposal of digester sludge. The WWTP process includes a three stage treatment of the wastewater (mechanical, biological, chemical) and digestion of the raw treatment sludge according to the technology mix in Switzerland. The functional unit of wastewater treatment is cubic meters of wastewater.

Treatment of wastewater with *variable composition* is inventoried. To achieve consistent inventories for different wastewaters, Excel calculation tools are created that calculate inventories from wastewater data. The model is essentially an extension of the model presented in (Zimmermann et al. 1996). Extensions include the consideration of total organic carbon TOC, dissolved organic carbon DOC, halogens and other elements as possible wastewater input. While in (Zimmermann et al. 1996) only biological oxygen demand BOD, chemical oxygen demand COD, nitrogen, sulfur, phosphorus and 12 metals were considered, in the present model an extended list of 39 elements in 53 compound or sum parameters are considered. Updates were performed for the generic sludge disposal options. The nitrogen flow through the WWTP was recaclulated.

The purification efficiency of treatment is not excellent. E.g. 95% of Swiss WWTP do *not* comply with the emissions threshold limit for ammonia set by Swiss legislation. The reason for this is lack of money in the municipalities for extensions of their WWTPs. Transfer coefficients for nitrogen compounds indicate that only 7 w% to 28 w% are removed from wastewater to air or to sludge, and the remainder is only transformed into other nitrogen compounds (mostly ammonia and nitrate), which remain in the water. Removal of carbon compounds is better and ranges at 80 w% to 90 w%.

Infrastructure of canalisation and the plants is differentiated into 5 classes depending on capacity and number of residential connections ranging from average capacity of 800 to 233'000 per-captia-equivalents (PCE). The inventory is targeted at *municipal plants* treating residential wastewater. This model cannot be used to inventory specialised wastewater treatment in *industrial processes*, except where it is similar to the municipal WWTP in technology and in the treated input.

Results showed the infrastructure of plants and sewers to be very relevant. In average operation, a very dilute pollutant charge is treated. But the accompanying water must be transported and processed too. This creates large infrastructure burdens.

Table of Contents

1 Introduction

The goal of this study is to create an Excel calculation tool to inventorise the treatment of different types of wastewater in a Swiss municipal wastewater treatment plants WWTP. The goal is to calculate *wastewater-specific* inventories for the treatment of diverse wastewater compositions. To achieve that, the energy and material flows through the plant during average operation are analysed, and the resulting outputs are allocated to the causing inputs.

This study is largely based on previous models created at ETH Zurich (Zimmermann et al. 1996). Important updates were performed on the nitrogen flow modelling in the WWTP, the sludge disposal choices, and the extension of the considered metal components.

2 Wastewater Generation and Treatment

2.1 Switzerland

In Switzerland 7'135'718 inhabitants in 2900 municipalities generated 1'441 million m^3 of wastewater in 2000 (BUWAL 2001b). Approximately 40'000 km of sewers are built (BUWAL 1997). Only a small share of connectable areas remain (1.8%), while 2.8% of the area cannot be connected to central WWTP (BUWAL 2001b). Most of the connectable but not connected inhabitants are located in the Canton of Ticino, where 16% of the population are not connected (BUWAL 2001b).

2.1.1 Wastewater Treatment Capacities

In 1965 in 14% of the Swiss population were connected to a municipal WWTP. In 2000 over 95% were connected (BUWAL 2001b). Approximately 40'000 kilometres of sewer pipes have been constructed. Switzerland had 979 WWTP in 1999. A detailed map of all Swiss WWTP is shown in (BUWAL 2002b).

WTTP capacities are expressed in **per-captia equivalents PCE**[1]. One PCE equates to a load of 60 grams of BOD in raw sewage per day, which is the typical BOD load generated by one person (BUWAL 1996:9). Swiss WWTPs treated 1441.5 million m^3 wastewater in 2000 (BUWAL 2001b) from 7'135'718 inhabitants, resulting in an annual amount of 202 m^3 of wastewater per PCE [2].

The Swiss WWTP are grouped in to different capacity classes (German 'Grössenklassen'). The majority of plants are small scale plants (class 4 and 5) for annual treatment of 10'000 PCE or less. However in these small plants only a minority of the Swiss wastewater is treated. Over 90% of the wastewater is treated in large plants (classes 1, 2 and 3).

Tab. 2.1 Data for the five capacity classes of WWTP in Switzerland 1993 (BUWAL 1994:22)

Capacity class	unit	1	2	3	4	5	total
Capacity range	PCE/a	over 100'000	50'000 to 100'000	10'000 to 50'000	2'000 to 10'000	30 to 2'000	20'110 [1]
Number of plants	unit	40	45	192	262	428	967
		4.14%	4.65%	19.86%	27.09%	44.26%	100%
		4.14%	8.79%	28.65%	55.74%	100.00%	–
Total treatment capacity	PCE /a	9'329'000	3'201'000	4'774'000	1'394'000	345'000	19'043'000
Share		48.99%	16.81%	25.07%	7.32%	1.81%	100%
Running sum		48.99%	65.80%	90.87%	98.19%	100.00%	–
Average capacity per plant in class	PCE/ a*unit	233'225	71'133	24'865	5'321	806	–

1 average capacity for a Swiss WWTP plant (falls in the class 3 range).

[1] also called 'inhabitant units' (German Einwohnergleichwerte EW or EGW).

[2] A similar value of 212 m^3 per captia and year is obtained if the 60 g BOD/captia*day are weighted with the average BOD concentration in untreated sewage of 103.6 g/m^3 (cf. Tab. 4.4 on page 14). In this study the value of 202 m^3 per captia and year is used.

2.1.2 Wastewater Treatment Technology

For an introduction to wastewater plant technologies and stages see chapter 3.3 'WWTP process description' on page 4ff. 68% of all Swiss WWTP (by capacity) have a tertiary stage (chemical treatment) with enforced precipitation of phosphate (GSDB 1994). 17% of all Swiss WWTP (by capacity) feature partial denitrification zones for elimination of nitrate (BUWAL 1996).

In Switzerland 15% of the connected areas have sewers that separate natural precipitation water from household wastewater (BUWAL 2001b). The natural precipitation water is returned to the natural water cycle.

Due to reduced heavy metal use, the metal content in wastewater treatment sludge has been declining since the mid-1980s (BUWAL 1997, 2002).

2.2 Other countries

The situation of connectivity and installed wastewater treatment technology in several OECD countries is shown in Fig. 2.1 (OECD 2001). The degree of development varies largely, e.g. between Netherlands, Switzerland and Sweden, where the over 90% of the population is connected to a secondary or tertiary stage WWTP versus Iceland, Turkey, Hungary, Mexico and Belgium, where plant connectivity is below 30% and over 70% of the wastewater is *not treated at all* (i.e direct emission).

The average OECD connectivity to WWTP in 1997 was 65%, with 9% being only connected to a mechanical treatment plant and 54% to a secondary (mechanical + biological) and or tertiary (mechanical + biological + chemical) treatment plant (OECD 2001). More on treatment plant technology in chapter 3.3 'WWTP process description' on page 4.

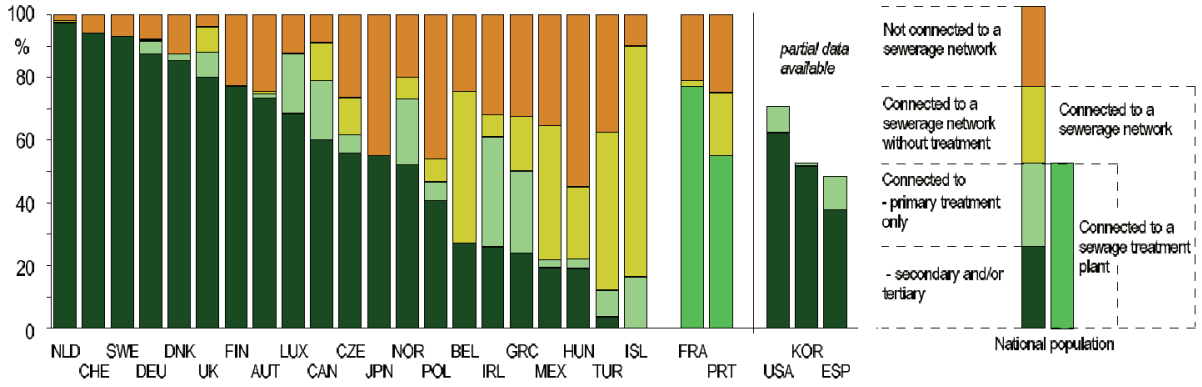

Fig. 2.1 Connectivity and installed wastewater treatment technology in OECD countries for late 1990s (OECD 2001). No data available for Italy, Australia and New Zealand.

3 Systems Characterisation

3.1 Scope

This study only inventories municipal wastewater treatment plants, that treat municipal sewage. Specialised WWTP for industries is inventoried in the inventory of these production processes and described in the respective reports.

3.2 Drainage and sewer system

Prior to the treatment of the wastewater in the WWTP, it is transported by sewer system from the producer to the plant. Within a building area, a residential drainage system collects wastewater and precipitation water and discharges them into the municipal sewer.

Hydraulic overload of the systems can lead to direct discharges of untreated wastewater from the sewer. These discharges represent direct emissions from the sewer system and are inventoried along with the wastewater treatment burdens. Chemical reactions of the wastewater within the sewer are not quantified and are neglected.

Wastewater can be altered by biological processes in the sewer, i.e. the wastewater discharged at the production location is not the same as the wastewater arriving at the WWTP. These processes are deemed to be similar to what is occurring in the biological stage of the WWTP and are neglected in this model.

3.3 WWTP process description

The treatment of wastewater can be differentiated into several steps, as the scheme in Fig. 3.1 shows.

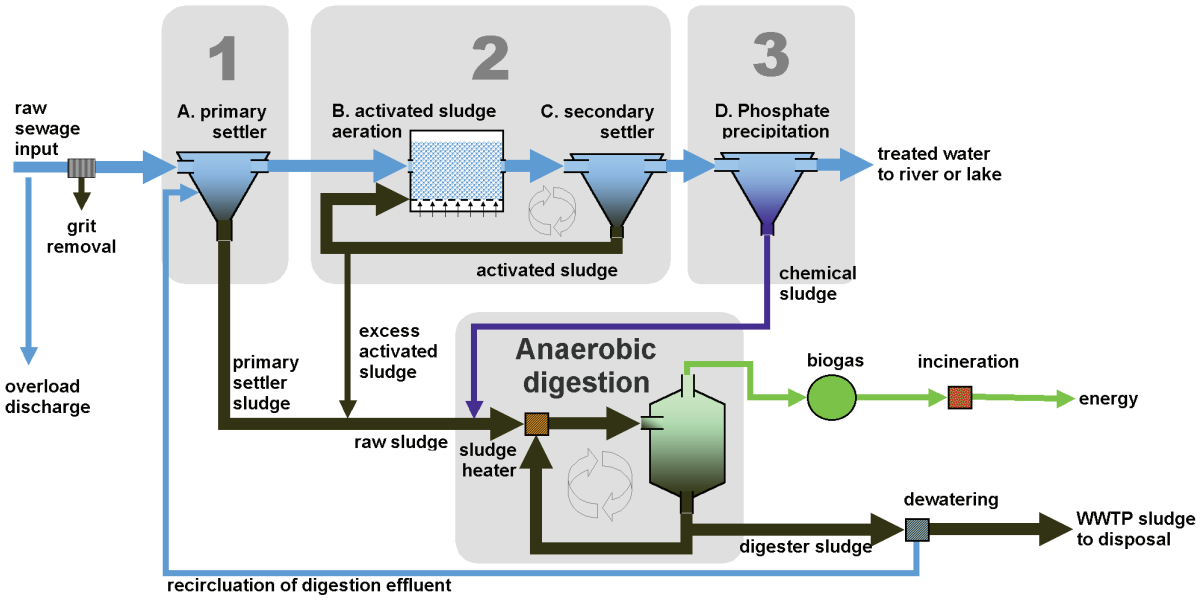

Fig. 3.1 Typical scheme of a three-stage wastewater treatment plant. Not to scale. Based on (Fritsche 1985:122).

Pre-treatment

Wastewater is collected by sewer system and led to the treatment plant site (top left in Fig. 3.1). The wastewater arriving at the WWTP is the raw sewage input. In times of hydraulic overload, e.g. due to

heavy rain, raw sewage can be averted from the WWTP. This overload discharge is emitted untreated to the receiving water body. Coarse material like branches, package waste is removed from the raw sewage input (grit removal).

Primary treatment (mechanical)

The most basic and most abundant treatment stage is a purely mechanical settling in a settling tank (box 1 in Fig. 3.1). Solids in the sewage accumulate over a few hours at the bottom of the tank by gravity and are removed as 'primary sludge'.

Secondary treatment (biological)

In the second stage, the wastewater from the primary stage enters a biological treatment (box 2 in Fig. 3.1). Micro-organisms degrade wastes in an aerated tank. These micro-organisms are the principal performers and the most crucial component of a WWTP. A selection of some micro-organisms are shown in Fig. 3.2. The degradation processes are essentially the same that take place in natural waters, but at a much accelerated rate and without exposure of the natural ecosphere to the untreated pollutants. The residence time is several hours. The aerobic environment converts ammonium NH_4^+ to nitrate NO_3^- (nitrification) and organic carbon to CO_2. Growth of micro-organisms leads to an accumulation of biomass in this stage. This biomass is present as sludge, which is separated by a second settling tank. This sludge is called 'activated sludge' since it stems from the biologically active stage. The activated sludge is recycled to the aeration tank. Excess sludge is separated. Some WWTPs feature also enforced denitrification in the secondary stage. In denitrification nitrate (NO_3) is microbially transformed to elemental nitrogen (N_2) which is emitted to air. Also some gaseous nitrous oxide N_2O is formed.

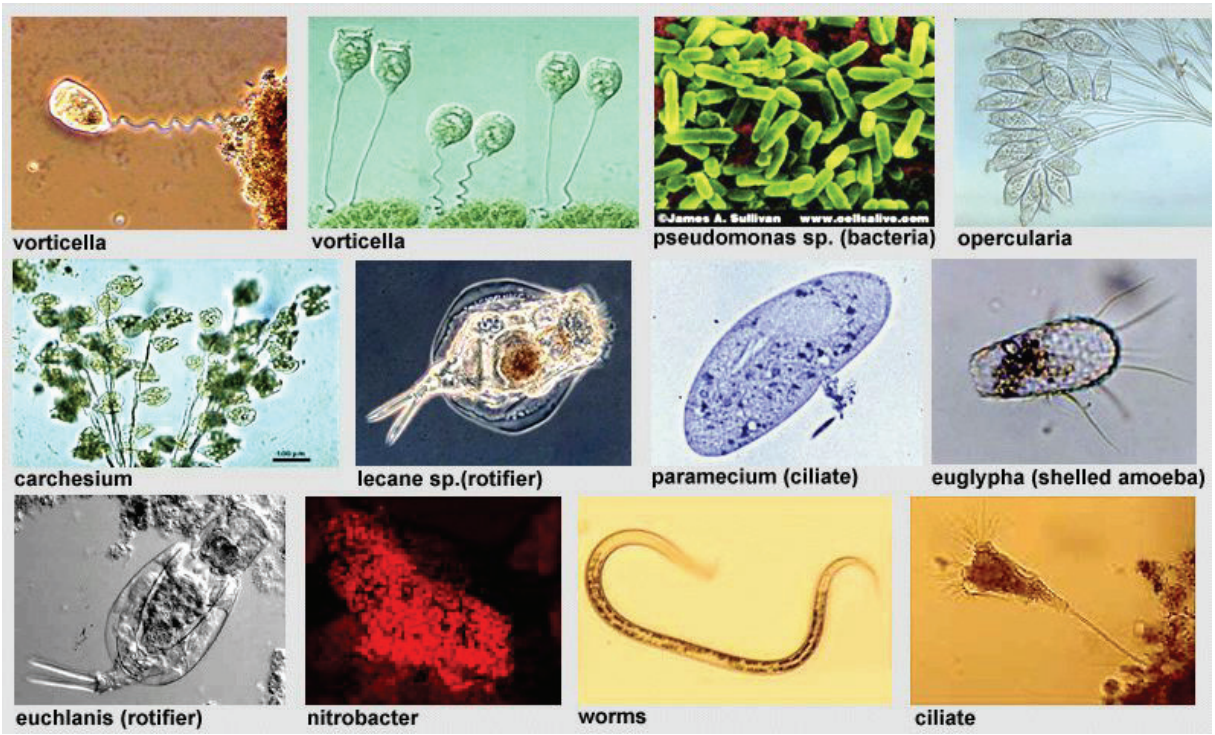

Fig. 3.2 **An arbitrary collection of micro-organisms present in activated sludge beds of WWTPs. At various scales (Crinum 2002, Engitech 2002, Hyperion 2002).**

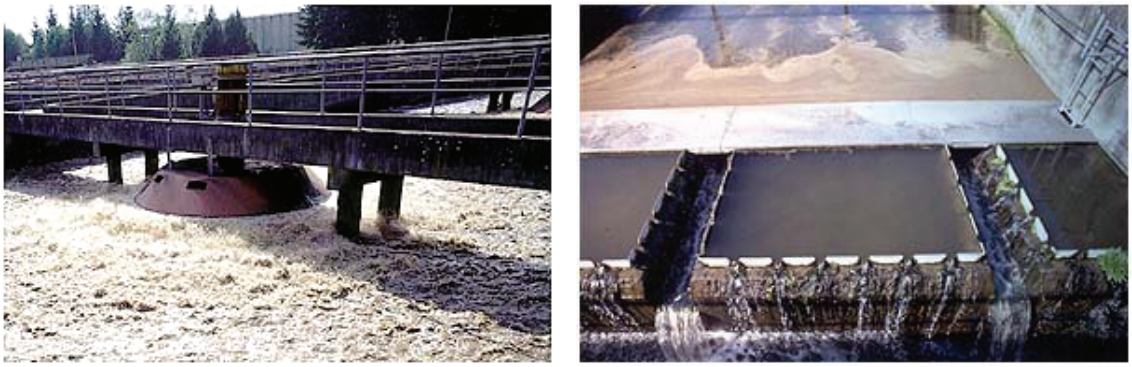

Fig. 3.3 **Activated sludge bed (left) and secondary settling tank (right) of WWTP Luzern (GALU 2001)**

Tertiary treatment (chemical)

Precipitation agents like iron chloride are added at this stage to precipitate phosphate as iron phosphate $FePO_4$. The solids from this stage are collected as chemical sludge. The treated water is ultimately discharged into a natural water body, usually a river. The treated water still contains pollutants and usually does not meet drinking water standards, but is deemed fit for discharge in a natural water, where further natural purification occurs.

Sludge digestion

At all three stages of wastewater treatment sludges are collected. These sludges (raw sludge) are further degraded by anaerobic digestion in a closed digestion tank over several days. Sludge flow is recycled and heated to optimise degradation. Biogas rich in methane (CH_4) can be obtained from this

process and incinerated for energy. The remaining sludge (digestion sludge) is de-watered and must be disposed off-site either as fertiliser, in landfills or in waste incinerators.

Currently sludge is generally digested in Swiss WWTPs[3]. Digestion and energy conversion infrastructure is rather expensive and has high maintenance demand. It might become more economical in the future to incinerate sludge without prior digestion. This development depends on local conditions regarding cost of sludge incineration, cost of digestion & energy conversion, and cost of commercial energy consumption The financial gain from energy production is usually small for WWTPs, since usually all energy is consumed internally and energy prices are low[3].

Fig. 3.4 Sludge digester tanks of WWTP Luzern (GALU 2001)

3.4 Capacity classes in the inventory

Former studies have shown that the different capacity classes of WWTP and sewers can show significantly different LCA results, depending on the impact category (Zimmermann et al. 1996). Treatment in a class 5 WWTP demands more infrastructure per m^3 of wastewater than a class 1 WWTP. Class 5 WWTP are usually located in more sparsely inhabited regions, leading to more sewer infrastructure necessary per captia.

For his reason the five capacity classes are distinguished in the inventory. This influences the infrastructure for the sewer grid and infrastructure for the WWT plant itself. The suggested default disposal pathways for digested sludge are differentiated according to capacity class.

3.5 Disposal of WWT sludge

Disposal of municipal WWT sludge in agriculture used to be common in Switzerland. In 1974 of a total of 90'000 tons of sludge, 80% was disposed as agricultural fertiliser (Stadelmann et al. 2002). In recent years demand has been declining, due to enforced nutrient budgeting on farms in accordance with integrated or organic farming rules, as well as concerns regarding soil protection as well as organic and inorganic pollutants in the sludge. Heavy metal contaminations have been steadily declining in WWT sludge (Stadelmann et al. 2002). In Switzerland agricultural disposal of WWT sludge is prohibited beyond January 2003 on cattle feed and vegetables production lands, and beyond October 2005 on all other agricultural areas (BUWAL 2002a).

[3] Personal communication with Max Maurer, EAWAG Dübendorf, March 5, 2002.

Fig. 3.5 Spreading of digested WWTP sludge on agricultural fields (landfarming). Pictures from (BUWAL 1997, BUWAL 2002a).

The utilised disposal options for WWT sludge in Switzerland for 2000 are shown in Tab. 3.1. Modelling of sludge disposal for the inventory is presented in chapter 5 'Sludge disposal' on page 37ff.

Tab. 3.1 Disposal of WWT sludge in Switzerland 2000 (BUWAL 2001a)

Disposal of WWT sludge	t/a	w-%
Total agriculture	78'357	38.6%
Liquid fertiliser	71'838	35.4%
Compost or granulate	6'519	3.2%
Total incineration	119'566	59.0%
Municipal waste incineration	21'592	10.6%
Cement kiln	33'683	16.6%
Special sludge incinerator	64'291	31.7%
Landflling	4'834	2.4%
Total	202'757	100.0%

In other European countries recycling in agriculture and landfilling is dominant. Recycling in agriculture is generally increasing and reduces landfilled masses. Spain and United Kingdom also discharge sludge directly to surface waters[4] (EEA 2002).

3.6 System boundaries

The goal of this study is to inventory disposal of different waste waters in a WWTP, so that this data can be used in LCAs of waste-producing activities. Average Swiss technology is assumed. Fig. 3.6 shows the inventoried process chain for wastewater treatment. Included processes are: transport in sewers, treatment in WWTP, sludge digestion, sludge disposal. Sludge disposal is a combination of application in agriculture (surface spreading) and incineration. For incineration in municipal solid waste incinerators the model described in part II is used. For landfilling of solid incineration residues the model described in part III is used. All other parts including application in agriculture are described in this part (IV).

[4] Presumably marine waters.

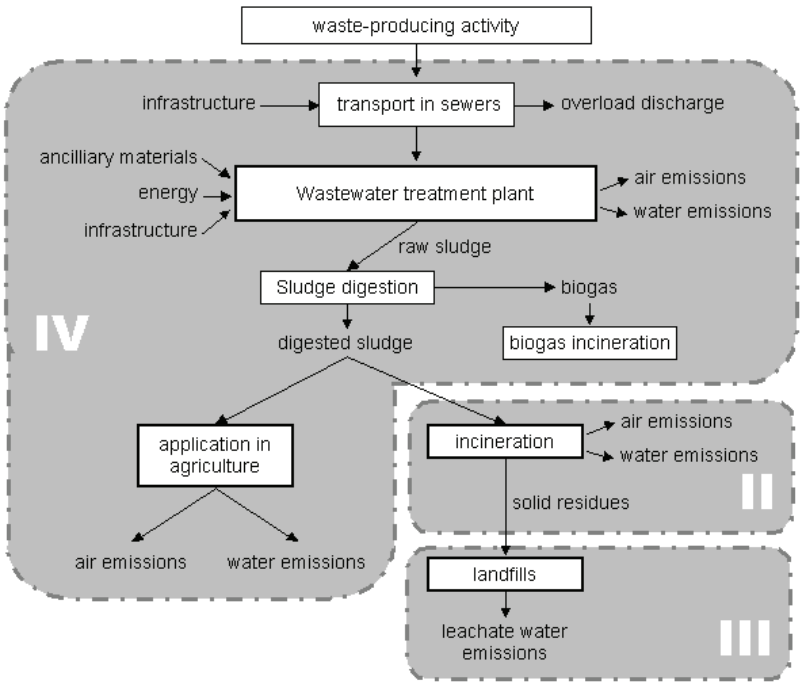

Fig. 3.6 **System boundaries for the wastewater treatment process chain. Roman numerals indicate the part of the report, where the applied disposal model is described.**

4 Life-Cycle Inventory of Wastewater Treatment

Remark: To avoid cluttered labels – like e.g. PO_4^{3-}–P for phosphorus in phosphate – all water pollutant species are given *without their ionic charge*. E.g. the phosphate anion (PO_4^{3-}) is denoted as PO_4. Likewise, for nitrate (NO_3), nitrite (NO_2) ammonium (NH_4), metal cations (e.g. Pb), other anions (e.g. Cl) etc.

4.1 Sewer system

For different WWTP capacity classes, different amounts for sewer infrastructure result. Diameters and pipe lengths in the grid vary. Tab. 4.1 gives the length of sewer per connected captia and the connected captia in the capacity class.

The residential sewer system is the sewer part located on an individual building area and is not owned by the municipality. Depending on the system boundaries, this sewer part can be added to the inventory by the user. By default it is not included in the inventory.

Tab. 4.1 Data for the sewer system grids and demand per m^3 wastewater.

Sewer grid capacity class	Meters per captia (Zimmermann et al. 1996:C.15)	Average PCE in class per plant (BUWAL 1994:22)	Million m^3 per lifetime of grid Column 2 * 202 m^3/a * 100 a	Grid length in km Column 1 * Column 2	Kilometer per m^3 wastewater Column 4 / Column 3
1	2.5	233'225	4'711	583	$0.1238 \cdot 10^{-6}$
2	3.4	71'133	1'437	242	$0.1683 \cdot 10^{-6}$
3	4.4	24'865	502.3	109.4	$0.2178 \cdot 10^{-6}$
4	5.7	5'321	107.5	30.3	$0.2822 \cdot 10^{-6}$
5	7.6	806	16.28	6.13	$0.3762 \cdot 10^{-6}$
residential	5.8	–	–	0.087	$0.2871 \cdot 10^{-6}$

The inventories in (Zimmermann et al. 1996) are based on (Labhardt 1996) and are given per m^3 wastewater. In ecoinvent 2000 the sewer infrastructure has the unit 'kilometer'. To convert the figures from (Zimmermann et al. 1996) in 'per m^3' to 'per kilometer', they are multiplied by the amount of wastewater generated by the number of connected PCE in the grid over the whole sewer lifetime and divided by the length of the grid in kilometer[5] (data cf. Tab. 4.1). One PCE equates to 202 m^3/a. A total lifetime of 100 years is used (Zimmermann et al. 1996:C.15).

Expenditures for renovation after 70 years are included in the infrastructure figures. Dredging of solid residues from the sewers is not inventoried. Most of dredged material is road waste washed to the sewers during rain and illegally disposed cat litter sand from private households. The sewers use gentle slopes and gravity to transport the wastewater, so no energy for pumps is needed.

The sum of concrete, gravel and cement is disposed using a dismantling module without recycling. Plastics and rubber are disposed in municipal waste incinerators. Steel and iron parts are assumed to be recyclable.

Default ecoinvent transportation distances and modes are used for all materials. Steel, cast iron is transported 600 km by train and 50 km by lorry. Plastics and rubber are transported 200 km by train and 50 km by lorry. Cement is transported 100 km by train and 20 km by lorry. Sand is transported

[5] Actually this calculation is independent of PCE, i.e. is identical to multiplying by the lifetime (100a) and the annual per captia wastewater generated (202m^3), and dividing by the specific grid length per captia in km.

50 km by lorry. Concrete and gravel is transported 20 km by lorry. Waste to waste incinerators is transported 10 km.

As relevant land use, only occupation as building site during the construction phase is occurring. Sewer pipes are usually built beneath roads. This type of occupation is typically only of short duration due to traffic requirements. An allocation question arises, if the land occupation during the road construction, shall be allocated between the different functions of the construction site. The function of a road construction site is providing infrastructure for road traffic but also infrastructure for any combination of other services: for sewers, for telecommunications connections, for television cables services, for electricity cables etc. The land occupation and transformations of road infrastructure are already inventoried in the road infrastructure; see inventories for road transport (Spielmann et al. 2003). In that report, no allocation was performed between the various mentioned functions of a road. For reasons of consistency and to avoid double counting no land occupation is inventoried here for sewer infrastructure.

Tab. 4.2 Life cycle inventory for 5 different capacity classes of municipal sewer grids and one residential grid.

Per km	location	Infrastructur	unit	class 1	class 2	class 3	class 4	class 5	residential
tap water, at user	RER	0	kg	22'090'000	18'940'000	15'780'000	13'410'000	11'050'000	3'156'000
concrete, sole plate and foundation, at plant	CH	0	m3	473.7	463.2	447.5	427.8	399	263.6
gravel, unspecified, at mine	CH	0	kg	838'000	828'500	790'600	748'000	698'300	319'600
sand, at mine	CH	0	kg	464'000	464'000	445'000	428'500	397'700	201'200
cement, unspecified, at plant	CH	0	kg	149'700	146'200	142'400	138'500	134'500	26'040
polyethylene, HDPE, granulate, at plant	RER	0	kg	43'300	38'300	33'300	28'900	24'260	19'330
polyvinylchloride, at regional storage	RER	0	kg	1'894	1'894	1'657	1'657	1'420	7'102
excavation, hydraulic digger	RER	0	m3	8'316	6'943	5'447	4'138	2'968	6'451
cast iron, at plant	RER	0	kg	8'759	8'759	8'759	8'759	8'759	5'129
diesel, burned in building machine	GLO	0	MJ	30'960	30'160	29'350	28'750	28'150	42'210
electricity, medium voltage, at grid	CH	0	kWh	18'780	16'550	14'290	12'420	10'360	5'885
reinforcing steel, at plant	CH	0	kg	51'840	51'840	51'840	51'840	51'840	-
chromium steel 18/8, at plant	RER	0	kg	17'990	17'990	17'990	17'990	17'990	-
polypropylene, granulate, at plant	RER	0	kg	1'894	1'894	1'894	1'894	1'420	-
synthetic rubber, at plant	RER	0	kg	473.4	473.4	473.4	473.4	473.4	-
transport, lorry 28t	CH	0	tkm	90'870	89'500	85'950	82'160	76'790	39'670
transport, freight, rail	RER	0	tkm	71'640	70'290	68'860	67'590	66'120	10'970
disposal, building, concrete, not reinforced, to final disposal	CH	0	kg	2'030'000	1'994'000	1'918'000	1'828'000	1'711'000	925'600
disposal, polyvinylchloride, 0.2% water, to municipal incineration	CH	0	kg	1'894	1'894	1'657	1'657	1'420	7'102
disposal, polyethylene, 0.4% water, to municipal incineration	CH	0	kg	45'200	40'190	35'190	30'790	25'680	19'330
disposal, rubber, unspecified, 0% water, to municipal incineration	CH	0	kg	473.4	473.4	473.4	473.4	473.4	-

4.1.1 Uncertainty for sewer infrastructure

Uncertainty for sewer infrastructure is estimated with the Pedigree approach (Tab. 4.3).

The uncertainty in energy consumption (diesel & electricity) is set equal to the uncertainty in material demand. This although the Pedrigree approach suggests smaller uncertainties for energy consumption

(basic uncertainty of 105%). For energy demand in building activities it seems unlikely that the demand in materials has a basic uncertainty of 300%, but the energy used to process and handle those materials has only a basic uncertainty of 105%. Consequently, energy consumption for infrastructure has the same basic uncertainty as the materials (300%). The same reasoning applies to disposal services. If infrastructure materials with an input basic uncertainty of 300% are disposed, the basic uncertainty cannot change to 105% for the disposal of those materials (as suggested by the Pedigree approach for waste disposal processes).

Uncertainty for transport services is calculated from the uncertainty in the transported masses using the Wilkinson-Fenton approximation for sums.

Tab. 4.3 GSD^2 value for sewer system infrastructure

Exchange	GSD^2 value	Pedigree scores	Comment
Infrastructure sewer system	305%	(2,2,2,1,1,5)	Basic uncertainty of 3; Data from one inventory study (Labhardt 1996)

4.2 Overload discharge

Hydraulic overload of the systems can lead to direct discharges of untreated wastewater from the sewer or WWTP reservoirs. It is estimated that in such discharges 1% of the particulate contents of the wastewater and 2% of the dissolved contents are emitted to river water. The value for particulate emission is lower due to sedimentation (Labhardt 1996).

The generic ratio of dissolved vs. particulate contents is either clear from the definition of the pollutant parameter or shown in Tab. 4.6 on page 17 and in Tab. 4.19 on page 25.

Apart from overload discharge no emissions from the sewer or transformation in the sewer – e.g. from the sewer slime (German 'Sielhaut') – is heeded in the model.

4.3 Wastewater purification process

The wastewater treatment has two major parts (cf. Fig. 3.1 on page 4):

1. Three-stage-purification of the wastewater
2. Digestion (fermentation) of the raw sludge resulting from step 1

The wastewater purification is described in this chapter. Digestion of the raw sludge is described in chapter 4.4 'Sludge digestion process' on page 28. The figures for effluent concentrations are for the complete process (i.e. heed the water from the digester recycled back to the WWTP input).

4.3.1 Transfer coefficients in wastewater purification

The contents of the raw sewage are physically and chemically altered in the three treatment steps of a WWTP, to purify wastewater or make its contents less harmful. Phosphorus shall be removed as far as possible. Ammonium NH_4 shall be converted to elemental nitrogen N_2 or nitrate NO_3. In this chapter, transfer coefficients are devised based on the *average* operation in WWTPs. These average transfer coefficients can then be used to calculate the outputs of a WWTP for a *specific, user-defined* wastewater.

The model is not differentiated according to capacity classes of the WWTP. Lack of data is a reason for this choice, but also indications that WWTP performance is not determined by capacity class as such, but more by applied technology. For example, nitrogen reduction is almost identical for different

capacity classes (BUWAL 1996). This means that the inventories of the wastewater treatment in the different capacity classes only differ in infrastructure[6].

Tab. 4.4 Concentration of pollutants in average WWTP input and output

	Average wastewater input g/m^3	Source	Average WWTP effluent g/m^3	Source
BOD	103.6	DBGS 1994	8.29	[1]
COD	155.6	[6]	27.97	[1]
DOC	45.75	[8]	6.52	[1]
TOC	67.3	[7]	6.52	[1]
NH_4-N	14.95	DBGS 1994 [2]	8.75	BUWAL 1996
NO_3-N	1.05	DBGS 1994 [2]	11.12	BUWAL 1996
NO_2-N	0.4	Zimmermann et al. 1996:C.40 [3]	0.2	Zimmermann et al. 1996:C.40 [4]
$N_{part.}$	3.28	[1]	0.5	BUWAL 1996:19
$N_{org.solv.}$	8.39	[1]	0	[5]
PO_4-P (dissolved)	2.459	[1]	1.273	[1]
P particulate	0.615	[1]	0	[1]
P total	3.0737	[1]	1.273	[1]
SO_4-S (dissolved)	44	Koppe 1993	44	Koppe 1993
S particulate	2	Koppe 1993	0	Koppe 1993
S total	46	Koppe 1993	44	Koppe 1993

1 calculated from model (see sections below)

2 50% of the original NO_3-N figure was converted to NH_4-N, based on a personal communication with Max Maurer, EAWAG Dübendorf, May and April 2002.

3 based on (AGW 1993)

4 based on personal communication with Markus Thomann, Institute for Hydroengineering, ETH Zürich, 1996.

5 personal communication with Max Maurer, EAWAG Dübendorf, May and April 2002.

6 derived from BOD using a generic COD/BOD ratio of 1.5 (Gujer 1993).

7 derived from BOD using an observed TOC/BOD ratio of 0.65 (UAG 1999).

8 derived from TOC using an observed DOC/TOCratio of 0.68 (100%-32% from Tab. 4.6).

The concentration of pollutants in input and output water during average operation of an average Swiss WWTP is shown in Tab. 4.4. These figures originate from the indicated literature source or from calculations, which are discussed in detail in the sections below.

For some pollutants the purification efficiency seems rather poor. This is not an error, but reflects the actual circumstances in recent average Swiss WWTPs. For example, the permitted threshold limit value for ammonia (NH_4-N) in WWTP output is 2 mg/l (GSchV 2001), while the actual average value is 8.75 mg/l, or 4.4 times above the threshold value. Approximately 96% of all Swiss WWTPs do not comply with the legal limit for ammonia, and approximately 85% of all wastewater is treated in non-compliant WWTPs (calculated from BUWAL 1996). This deficiency has been known for years, but many municipalities struggle with finding the monetary resources for maintenance, extension or re-design of their WWTP (BUWAL 1997:59). Since the legal limits are not of any relevance in defining the status-quo for this inventory study they are not reproduced here. The interested reader is referred to the relevant legal text (GSchV 2001).

[6] The capacity class of a WWTP also determines the generic *default* disposal path for digester sludge in the calculation tool, which can be overridden by the user. In that respect, capacity class can have an additional influence on the inventory.

The input data for an *individual* plant can differ from the *Swiss average* as detailed in Tab. 4.5. This difference is partly covered by the uncertainty in the sewage input, which can vary for different regions or different settlement structures[7]. Differences in concentrations also depend on the degree of meteor water exclusion from the plant which can differ locally. However, the deviation of individual plants is naturally larger than the uncertainty rage for the average Swiss sewage. Differences in the *output* data are due to different input and different technology (e.g. apparently a righteous phosphate elimination in WWTP Bern).

The data for Swiss average is mostly based on a large sample of Swiss WWTPs (DBGS 1994, BUWAL 1996). For some parameters hundreds of different plants are heeded in these sources. Currently more than 970 Swiss WWTPs are in operation. Regarding data quality it seems more accurate to use data from a *large sample*, than to use the new data of *only a few individual plants*[8]. The advancement in WWTP technology is not fast, as most municipalities lack resources even to *maintain existing* WWTP technology (BUWAL 1997:59). The difference between performance data of the mid-90ties and 2000 is therefore not likely to be large[9]. The mid-90ties data, backed by a large sample, seems to be preferable to newer data of only a few reporting WWTPs, which are usually modern and large facilities. Otherwise not a Swiss average would be modelled, but in tendency the large, well-equipped fraction of WWTPs would be over-represented.

7 The uncertainty for water emissions with the Pedrigree approach is larger or equal to 150%; ≥ 500% for metals.

8 The *new* BUWAL WWTP database (DBGS II) is currently in preparation at BUWAL within the project GEWISS (water information system) of Federal Office for Water and Geology FOWG and BUWAL (BUWAL 2002c).

9 Personal commmunication with Max Maurer, EAWAG Dübendorf, February 2003.

Tab. 4.5 Comparison of individual Swiss plants data with average Swiss data

in mg/l	Aargau 1998	Kloten 2001	Bern 2001	Data used for all plant classes
Description	All 81 WWTPs of Canton Aargau [2]	1 WWTP of class 2	1 WWTP of class	Swiss average.
Source	(UAG 1999)	(Liebi 2002)	(ARB 2002)	Taken from Tab. 4.4
Sewage input	120 Mio. m^3/a	7.8 Mio. m^3/a	60 Mio.m^3/a	1441.5 Mio. m^3/a
BOD input	162		250	103.6
BOD effluent	10.2		12	8.29
COD input		257	439	155.6
COD effluent		13.8	40	28
TOC input	105			67.3
TOC effluent	10.2			6.52
NH_4-N input	17.5		17	14.95
NH_4-N effluent	5.75		16	8.75
NO_3-N input				1.05
NO_3-N effluent			2.27	11.12
NO_2-N input				0.4
NO_2-N effluent			0.22	0.2
N_{tot} input		36		28.07
N_{tot} effluent		17.8	21	21
PO_4-P input				2.459
PO_4-P effluent			0.02	1.27
P_{tot} input	3.25	4.49	6.01	2.459
P_{tot} effluent	0.917	0.513	0.26	1.273

Blank cells indicate not reported figures.

1 2 class-1 plants, 6 class-2 plants, 19 class-3 plants, 31 class- 4 plants, 20 class-5 plants, 3 plants no data.

Transfer coefficients for carbon compounds

The overall development of carbon compounds in wastewater systems is usually monitored using the sum parameters BOD, COB, DOC and/or TOC. More general information on these parameters in the section 'Abbreviations' on page 48.

A part of the carbon in WWTP is removed in the primary settler to raw sludge (see Tab. 4.6). Another part is removed in the activated sludge bed either by incorporation into the sludge biomass or by oxidation to CO_2 and subsequent air emission. Not all carbon is removed from the wastewater and some pollution remains in the effluent (see Tab. 4.7). With transfer coefficient data for two of the processes, the transfer coefficients for the remaining process (elimination in activated bed) can be calculated by difference (see Tab. 4.8). A fraction of 58%[10] of the carbon eliminated in the activated bed is transferred to the activated sludge and 42% are removed as CO_2 (see Tab. 4.8).

[10] Assumption from Zimmermann et al. 1996:C.25. Typical sludge fractions are in the range of 50% to 67% (Gujer 1995).

Tab. 4.6 Transfer coefficients to primary settler sludge for carbon compounds. Corresponds to the particulate share.

Transfer coefficient to primary settler sludge	Ottenbach ZH (Thum 1995)	Klagesheim (Gujer 1993)	Zürich city (Gujer 1993)	**This study**
BOD	26%	51%	–	**39%**
COD	31%	–	33%	**32%**
DOC	–	–	–	**0%** [1]
TOC	–	–	–	**32%** [2]

1 no elimination of dissolved carbon

2 assumed equal value as for COD

Tab. 4.7 Transfer coefficients to effluent for BOD and COD for carbon compounds

Transfer coefficient to effluent	Ottenbach ZH (Thum 1995)	Switzerland (DBGS 1994)	Zürich city (Gujer 1993)	Canton of Aargau (UAG 1999)	**This study**
BOD	6%	8%	–	–	**8%** [1]
COD	19%	–	18%	–	**18%** [1]
DOC	–	–	–	–	**14.2%** [2]
TOC	–	–	–	9.7%	**9.7%**

1 data from (Gujer 1993) and (DBGS 1994), representing extensive measurements, was adopted.

2 Set so that the DOC value in the output effluent equals the TOC value, i.e. all carbon is dissolved (6.52/67.3 = 9.7% from Tab. 4.4).

Tab. 4.8 Transfer coefficients for elimination in activated bed for carbon compounds

	Transfer coefficient elimination in activated bed (by difference)	of which as activated sludge 58%	of which as CO_2 42%
BOD	53.0%	30.7%	22.3%
COD	50.0%	29.0%	21%
DOC	85.8%	49.7%	36.0%
TOC	58.3%	33.8%	24.5%

Primary settler sludge and activated sludge are combined and discharged to the digester. E.g. 69.7% of the BOD input is eliminated as sludge (see Tab. 4.9).

Tab. 4.9 Transfer coefficients for carbon compounds in this study

	Transfer coefficient to sludge (sum of primary sludge and activated sludge) cf. Tab. 4.6 and Tab. 4.8	Transfer coefficient to air as CO_2 cf. Tab. 4.8	Transfer coefficient to effluent cf. Tab. 4.7
BOD	**69.7%** (= 39% + 30.7%)	**22.3%**	**8%**
COD	**61.0%** (= 32% + 29.0%)	**21.0%**	**18%**
DOC	**49.7%** (= 0% + 49.7%)	**36.0%**	**14.2%**
TOC	**65.8%** (= 32% + 33.8%)	**24.5%**	**9.7%**

Speciation of carbon emissions

All parameters BOD, COB, DOC and TOC relate to carbon compounds, i.e. represent partially overlapping parameters. The user has the possibility to define combinations of these parameters in the calculation tool.

Carbon from BOD, COD, DOC, TOC: For the calculation of the CO_2 air emissions and carbon mass to sludge digestion, the four parameters must be converted to one value for carbon mass. A reduction in BOD (ΔBOD) means that organic substance was removed and thus oxygen take-up in the BOD test will be smaller[11]. The difference in oxygen take-up (ΔBOD) can not simply be translated into a figure for carbon or CO_2. Not all of the carbon that was removed as CO_2 *in the WWTP* (leading to a reduction of BOD) is necessarily mineralised to CO_2 *in a BOD test*. I.e. the conversion factor for ΔBOD-to-carbon is larger than the equation $C + O_2 => CO_2$ suggests (12/(2*16)=0.375). To obtain a conversion factor for ΔBOD to carbon, the relation of ΔBOD and ΔTOC during the average operation of the WWTP is regarded (cf. Tab. 4.4). ΔBOD is 95.31 g O_2/m^3 and ΔTOC is 61.1 g C/m^3. From that a ΔBOD-to-carbon factor of 0.641 is derived. Similarly, for COD a ΔCOD-to-carbon factor of 0.479 is derived from ΔCOD is 127.63 g O_2/m^3 and ΔTOC is 61.1 g C/m^3. The ΔCOD-to-carbon factor is smaller than the ΔBOD-to-carbon factor since the probability that carbon can resist mineralisation in a COD test is smaller than in the BOD test. For the reasons stated above both factors are larger than the factor derived from chemical stochiometry (0.375).

For DOC and TOC no conversion is necessary, as the carbon removed to air is determined irrespective of whether it was soluble or in a particulate matrix. From the four parameters, *four* carbon mass values can be calculated. Of course, only *one* carbon value must be used in further calculation to avoid double counting. If the four extrapolated carbon values do no coincide, priority is given to the value derived from TOC (proper mass balance). If no TOC value was defined, the largest of the carbon values is used.

In ecoinvent 2000 emissions of CO_2, CO, and CH_4 are discerned in to emissions of biogenic origin and of fossil origin. The emissions of these pollutants occur during the biological stage of the wastewater treatment, the incineration of biogas from sludge digestion, and incineration of digester sludge in MSWI. All these carbon emissions are specified according to the carbon in the input wastewater, i.e. if the wastewater is condensate from light fuel oil boilers, all carbon emissions are classified as 'fossil'[12].

Since the CO_2 from WWTP originates from the metabolism of living microbes in the biological treatment, the resulting emission is classified as biogenic. Also the carbon in digested sludge

[11] BOD can also measure oxygen take-up *other than* from hydrocarbon mineralisation, most prominently from *nitrification* of NH_4 to NO_3. For this reason BOD is normally measured with inhibition of nitrification (GSchV 2001:33). The calculation procedure for water emissions in ecoinvent 2000 does not heed other oxidation than from hydrocarbon mineralisation (Jungbluth 2001, see also Frischknecht et al. 2007). Therefore, reduction in BOD relates entirely to a reduction of (hydro)carbon compounds leading to emissions of CO_2 and H_2O.

[12] This choice might be debatable, since emissions from the *metabolism of living microbes and the resulting biomass (sludge)* could generally classified as 'biogenic' – meaning 'from the living'. However, the *origin* of the carbon in the wastewater is considered here for classification. In the IPCC guidelines for national greenhouse gases biogenic and fossil emissions are also discerned. According to (Hobson et al. 2000:5.10&5.28), CO_2 from biogenic waste incineration and landfill gas incineration should not be counted in national greenhouse gas inventories (*'CO_2 emissions from landfill gas recovery combustion are of biogenic origin and should not be included in national totals.' & 'It can be difficult to differentiate between the biogenic and the fossil part of waste going for incineration.'*). On the other hand, CH_4 emissions from WWTPs are fully counted in national inventories irrespective of the origin of the carbon source in the wastewater, e.g. wastewater from beer production are apparently inventoried alike to wastewater from organic chemical production (Hobson et al. 2000:5.14ff.&5.22). CO_2 emissions from WWTPs are apparently neglected in national inventories; presumably on the grounds that they are negligible in a national perspective. However, in this study a perspective per plant is adopted, and the CO_2 emissions are not negligible.

(biomass) is classified as biogenic. This choice might be debatable, if carbon in the wastewater originates from fossil sources (e.g. condensate water from light fuel oil boilers).

Water emissions of BOD, COD, DOC, TOC: For the overload discharge emissions all parameters BOD, COB, DOC and TOC are inventoried. Gaps are filled by extrapolation with generic conversion factors derived from raw (untreated) wastewater, shown in Tab. 4.10. For the WWTP water effluent all four parameters are inventoried. Gaps are filled by extrapolation with generic conversion factors derived from *treated* wastewater, shown in Tab. 4.11. If a parameter value is obtained directly from the model calculation this value is given priority over parameter values from extrapolations using generic conversion factors. If the extrapolated values do not coincide, the largest value is used. The four parameters are inventoried *in parallel* for all water emissions.

Tab. 4.10 Generic conversion factors relating to organic carbon content in untreated wastewater

Ratio in untreated wastewater	Generic value	Source
BOD/TOC	1.54	(UAG 1999)
COD/TOC	2.31	from BOD/TOC * TOC/COD = 1.54 * 1.5 (Gujer 1993).
DOC/TOC	0.68	dissolved part (cf. Tab. 4.6)

Tab. 4.11 Generic conversion factors relating to organic carbon content in treated wastewater

Ratio in treated wastewater	Generic value	Source
BOD/TOC	1.27	from model
COD/TOC	4.29	from model
DOC/TOC	1.00	assumption: all carbon is dissolved

Transfer coefficients for phosphorus

Phosphorus in wastewater is present as dissolved phosphate PO_4 or as particulate phosphorus (inorganic precipitates or in biomass). In the WWTP effluent, phosphorus is only present as dissolved phosphate (Tab. 4.12). Particulate phosphorus in the wastewater input is completely transferred to the sludge. Transfer of phosphorus to air is negligible. Not all WWTP feature enhanced phosphate precipitation. 32% of all WWTP do not have any enhanced phosphate precipitation (GSDB 1994). The remainder uses iron sulfate, iron chloride, aluminium sulfate and other agents to precipitate dissolved phosphate (Tab. 4.13). Precipitation enhances phosphor transfer to the sludge. Metal cations from the agents (Fe, Al) are also transferred to sludge. Anions from the agents (SO_4, Cl) are released to the effluent.

Tab. 4.12 Phosphor flows for WWTP with conventional third stage simultaneous precipitation of phosphate

	wastewater input g/m^3	in effluent g/m^3	to sludge g/m^3	transfer coefficient effluent	transfer coefficient sludge
PO_4-P	2.46 [1]	1.273 [2]	1.185	51.8%	48.2%
P_{part}	0.615 [1]	0 [3]	0.615	0.0%	100.0%
P_{total}	3.07 [1]	1.273	1.8	41.4%	58.6%

1 derived from (BUWAL 1996:25).

2 calculated from the measured P_{total} elimination in Tab. 4.14.

3 complete elimination of particulate phosphor assumed.

Tab. 4.13 Technology mix of enhanced phosphor precipitation in Swiss WTTPs (Zimmermann et al. 1996:C.33 based on DBGS 1994)

P-elimination	Shares (GSDB 1994)		Stochiometry
With $FeSO_4$	24%	27.09%	1
With $FeCl_3$	31%	34.68%	1
With $Al_2(SO_4)_3$	6%	6.50%	2
others/combinations	8% [1]		
none/n.a.	32%	32%	–
	100%	100%	

1 assumed to be distributed according to the first 3 entries

Precipitation agents have different ability to eliminate[13] phosphate based on their stochiometry (Tab. 4.13). Precipitation agents can also react with hydroxy anions. Therefore, all agents are added in stochiometric excess. For the common third stage simultaneous precipitation, the mol ratio of $FeSO_4$-equivalents to dissolved PO_4 in wastewater *input* is 1.5 (BUWAL 1996:25). With an PO_4 elimination of 64.8% for this technology (Tab. 4.12) this translates to 4.26 mols of $FeSO_4$-equivalents per *eliminated* mol PO_4.

Approximately 68% of all wastewater is treated in WTTPs featuring some form of enhanced phosphor precipitation (Tab. 4.13). Approximately 12.8% of all plants by capacity feature a flocculation-filtration stage, sometimes also referred to 'fourth stage' purification (BUWAL 1994:29). This data refers to 1991 but has not changed much since then[14]. Flocculation-filtration enhances precipitation. Phosphate elimination is approximately 96% in such plants (KfU 1992). Flocculation-filtration also makes more efficient use of precipitation agents by cascading precipitation. The two stage precipitation lowers agent use by a factor of 0.485 (= $\sqrt{4.26}/4.26$). The agent use in gram per eliminated gram of PO_4-P is shown in Tab. 4.14. The agent use is calculated from the eliminated amount of phosphorus. The agents are transported prior to consumption 600 km by train and 50 km by lorry (ecoinvent default distances).

13 'Elimination' here means transfer from the water to the sludge.

14 Personal communication with Max Maurer, EAWAG Dübendorf, May and April 2002.

Tab. 4.14 Precipitation technology data

Technology	No precipitation (GSDB 1994)	Simultaneous precipitation	Flocculation-filtration	average plant
Capacity of all plants	32%	55.5%	12.8% [1]	
PO_4-elimination	0%	64.8% [2]	96% [3]	**48.2%**
				g agent (100%) per g PO_4-P removed
Input $FeSO_4$	0	8.27	4.01	**7.47**
Input $FeCl_3$	0	11.31	5.48	**10.22**
Input $Al_2(SO_4)_3$	0	2.24	1.084	**2.02**

1 in 1991; from BUWAL 1994:29

2 UAG 1999

3 KfU1992:107

With the information on phosphorus concentration in input given in Tab. 4.4 and the transfer coefficient of PO_4 to raw sludge for the average technology mix, the average transfer coefficients for phosphorus can be calculated (Tab. 4.15).

Tab. 4.15 Calculation of average transfer coefficients for phosphorus during wastewater purification

	Average wastewater input g/m^3	Average WWTP effluent g/m^3	Transfer to sludge (difference) [1] g/m^3	Transfer coefficient effluent [1] g/g	Transfer coefficient sludge [1] g/g
PO_4-P (dissolved)	2.459 [1]	1.273	1.185	51.8%	48.2% [2]
P particulate	0.615 [1]	-	0.615	0.0%	100.0%
P total	3.0737 [1]	1.273	1.800	41.4%	58.6%

1 from Tab. 4.12.

2 from Tab. 4.14.

The user has the possibility to define phosphorus input concentrations as dissolved phosphorus in phosphate (PO_4-P), particulate phosphorus ($P_{part.}$), or total phosphorus ($P_{tot.}$). Data given as total phosphorus will be converted to a mixture of PO_4-P and $P_{part.}$ according to average operation data (see Tab. 4.4).

Transfer coefficients for sulfur

Sulfur in wastewater is present as dissolved sulfate SO_4 or as particulate sulfur (inorganic precipitates or in biomass). In the WWTP effluent, sulfur is only present as dissolved sulfate. Particulate sulfur in the wastewater input is completely transferred to the sludge. These is some negligible uptake of dissolved sulfur in the activated sludge, which is not quantified here.

Additional sulfur is found in the effluent, because some agents used to precipitate phosphate release sulfate ($FeSO_4$, $Al_2(SO_4)_3$). This contribution depends on the phosphate content of the wastewater and is not included here.

With the information on sulfur concentration in input and output given in Tab. 4.4, the transfer coefficients to effluent and to raw sludge can be calculated.

The user has the possibility to define sulfur input concentrations as dissolved sulfur in sulfate (SO_4-S), particulate sulfur ($S_{part.}$), or total sulfur ($S_{tot.}$). Data given as total sulfur will be converted to a mixture of SO_4-S and $S_{part.}$ according to average operation data (see Tab. 4.4).

Tab. 4.16 Calculation of transfer coefficients for phosphorus during wastewater purification

	Average wastewater input g/m^3	Average WWTP effluent g/m^3	Transfer to sludge (difference) [1] g/m^3	Transfer coefficient effluent [1] g/g	Transfer coefficient sludge [1] g/g
SO_4-S (dissolved)	44	44	0	100%	0.0%
S particulate	2	0	2	0.0%	100.0%
S total	46	44	2	95.7%	4.3%

1 calculated

Transfer coefficients for nitrogen

Nitrogen flow through a WWTP is complicated by the fact that nitrogen is present in many different forms that can be converted into each other. Reactions occur at different stages of the WWTP process. Following processes occur:

1. Sedimentation and transfer to raw sludge of originally particulate constituents or by absorption to activated sludge particles.
2. Nitrification: ammonium and nitrite NO_2 is oxidised to nitrate NO_3 by nitrifying micro-organisms.
3. Growth of biomass incorporates nitrogen
4. Denitrification: nitrate NO_3 is reduced to elemental nitrogen N_2 by micro-organisms and emitted to air.

Species measurements are not available for all stages. The flow of nitrogen through the three stages of the water purification of the WWTP is modelled to calculate the average transfer coefficients. The basis for the calculation are the concentrations of nitrogen species measured in input (raw wastewater) and outputs (effluent, sludge, digester gas). Unknown values are calculated by difference. The modelled nitrogen species are ammonium NH_4, nitrate NO_3, nitrite NO_2, dissolved organic nitrogen $N_{org,solv}$ and particulate nitrogen N_{part}.

A note of caution to the interested reader: the model essentially consists of four steps which are calculated consecutively, although in reality they occur partly in parallel. The calculation model is *recursive*. The recursive calculation is necessary, as in general the result parameters depend on unknown parameters, that are in turn defined by known (measured) parameters *and* the sought-after result parameters. Instead of attempting a formal solution of the resulting set of equations, the relations of the parameters are modelled in an Excel spreadsheet. A solution is then obtained by allowing recursive calculations in Excel. In the following the essential relations of the parameters in the model are described. The resulting figures are shown in Tab. 4.17.

All figures refer to one m^3 of wastewater input. All nitrogen species are given per weight of nitrogen, *not per weight of compound* (cf. Tab. 4.4) The water volume output is assumed approximately equal to the input. The water removed with the sludge is <0.2%, i.e. negligible.

I. Primary settler

The primary settler removes solids from the wastewater. In this step the particulate nitrogen N_{part} is completely removed and transferred to the sludge. The efficiency of removal is assumed to be 100%. The total raw sludge generated in the WWTP is 125g per m^3 of wastewater input (DBGS 1994). 52.5% of that sludge is generated in the primary settler (= 32%/61%, cf. Tab. 4.9 on page 17), the rest in the secondary settler (removal of activated sludge, see step III below). Assuming a nitrogen content in the primary sludge of 5% the nitrogen removed in the primary sludge is 3.28 g per m^3. This nitrogen

originates exclusively from particulate nitrogen in the wastewater input, i.e. no other process contributes to the nitrogen in primary sludge.

II. Nitrification

In the activated sludge bed, ammonium NH_4, nitrite NO_2 and dissolved organic nitrogen $N_{org,solv}$ are partly converted to nitrate NO_3. Some of that nitrate is reduced to elemental nitrogen in the denitrification (see step IV below). Some nitrate is also taken up by the activated biomass and is transferred to the sludge (see step III below). This defines the amount of NO_3 that must be generated in this step. Following processes contribute to the generation of nitrate:

1. Nitrification is the only step that reduces nitrite. The nitrification of nitrate can therefore calculated directly from the difference between measured nitrite in the wastewater input and the effluent.

2. Dissolved organic nitrogen $N_{org,solv}$, which is usually present in amines and urea, is converted to NH_4 and from that to NO_3. It is assumed that this chain of conversions occurs with the same rate as the conversion of NH_4 in the input to NO_3 (cf. process 3 below).

3. The third process contributing to NO_3 formation is conversion of NH_4 in the input to NO_3. This amount can be calculated by difference from the total amount of NO_3 that must be generated by nitrification minus the contribution of the other two processes (from nitrite and dissolved organic nitrogen).

III. Buildup of activated sludge (biomass)

In the next step, nitrogen in amonium NH_4, nitrate NO_3 and dissolved organic nitrogen $N_{org,solv}$ is partly incorporated in the biomass of the activated sludge. Excess sludge is removed to the digester in the secondary settler. A part of this biomass is emitted as particulate nitrogen in the effluent. As all of the particulate nitrogen in the input was removed completely in the primary settler, the particulate nitrogen in the effluent must completely stem from this process. 47.5% of the total sludge (125 g/m^3) is secondary settler sludge (= 29%/61%, cf. Tab. 4.9 on page 17). Secondary sludge contains 4.5% nitrogen (Gujer 1993). Hence, 2.67 g N per m^3 is transferred to secondary settler sludge. Additionally, 0.5 g N per m^3 is released as particulate nitrogen in the effluent. Therefore, a total of 3.17 g N per m^3 is formed from NH_4, NO_3 and $N_{org,solv}$. It is assumed that 10% of that amount is formed from NO_3, 9% from $N_{org,solv}$ and the remaining 81% from NH_4.

IV. Denitrification

In denitrification, nitrate NO_3 is reduced to elemental nitrogen N_2 by micro-organisms. A minor fraction is converted to nitrous oxide N_2O. Denitrification is dependent on the operating conditions in the WWTP (BUWAL 1996). Plants without continuous nitrification (type A) convert 3.2% of the total nitrogen input to a direct air release of N_2 and N_2O. Plants with continuous nitrification, but no denitrification (type B) reduce nitrogen by 12.9%. Plants with a denitrification zone in activated sludge bed (type C) reduce nitrogen by 32.3%. The capacity-weighted technology mix of all types is 72.65% type A, 13.48% type B, and 4.41% type C, respectively. The capacity-weighted total denitrification rate relative to the total nitrogen input is hence 5.5%. The denitrification rate to N_2O is 0.02%-0.07% relative to the total nitrogen input [15]. A geometric mean of 0.037% is used in the model (i.e. 0.68% of the denitrified nitrogen to air is released as N_2O).

There is a gap between the ammonium present after the calculations in the third step (biomass build-up) and the ammonia in the effluent. To close this gap, it is assumed that dissolved organic nitrogen is converted to ammonium.

Tab. 4.17 Modelled nitrogen flow through WWTP. Arrows indicate transfers and reactions

		I		II		III		IV		
	average Inflow	Reduction Sedimentation	after Sedimentation	Reduction nitrification	after nitrification	Biomass buildup	after biomass buildup	Reduction denitrification	after denitrification	average outflow
	g N/ m3	g N/ m3	g N/ m3	g N/ m3	g N/ m3	g N/ m3	g N/ m3	g N/ m3	g N/ m3	g N/ m3
NH4-N	14.95		14.95	7.495	7.455	2.571	4.884	-3.867	8.751	8.751
NO3-N	1.05		1.05	-11.9	12.95	0.2857	12.67	1.545	11.12	11.12
NO2-N	0.4		0.4	0.2	0.2		0.2		0.2	0.2
N part.	3.279	3.279				-0.5	0.5		0.5	0.5
N org. solv.	8.392		8.392	4.207	4.185	0.3174	3.867	3.867		
total N	**28.07**		**24.79**		**24.79**	**2.674**	**22.12**	**1.545**	**20.57**	**20.57**
removed to sludge		3.279				2.674				
removed to air								1.545		

With the nitrogen flow model completed, the transfer coefficients describing the fate of individual substances can be calculated. This consist of tracing every feasible reaction path for a certain input (e.g. NH_4 is partly converted to NO_3 which is partly converted N_2) and establishing all the resulting reaction products. In steps where a substance concentration *increases* from reactions from other substances (e.g. NO_3 in nitrification, step II) no increase for this substance must be used (since 1 gram of NO_3 in the input cannot produce more than 1 gram of NO_3). Summing up over all different reaction products for all reactions pathways results in the transfer coefficients.

[15] Personal communication with Max Maurer, EAWAG Dübendorf, May and April 2002.

Tab. 4.18 Transfer coefficients for nitrogen species

input as:							
NH_4-N	32.7%	43.0%		2.9%		15.4%	6.0%
NO_3-N		85.9%		0.3%		1.9%	11.9%
NO_2-N		42.9%	50.0%	0.2%		0.9%	6.0%
$N_{part.}$						100.0%	
$N_{org. solv.}$	46.1%	43.0%		0.8%		4.1%	6.0%
SKN	37.5%	43.0%		2.1%		11.4%	6.0%
TKN	32.9%	37.7%		1.9%		22.3%	5.2%
output as:	NH_4-N in water	NO_3-N in water	NO_2-N in water	$N_{part.}$ in water	$N_{org. solv.}$ in water	sludge-N to sludge	N_2, N_2O to air

In the model all wastewater input data relating to soluble Kjeldahl nitrogen SKN, total Kjeldahl nitrogen TKN or total nitrogen are converted to values for NH_4-N, NO_3-N, NO_2-N, $N_{part.}$, and $N_{org.solv.}$ using the concentrations in average wastewater (see Tab. 4.4). These extrapolated values are used to fill data gaps, if the user has not provided these parameters, i.e. direct data provided by the user is used with priority.

Transfer coefficients for other elements

Metals have various elimination rates in the WWTP, i.e. the removal efficiency from wastewater is diverse. Removed metals are transferred to the sludge. Essentially the elimination reflects how much of the element is particulate and how much is dissolved in the wastewater. The values in Tab. 4.19 are taken from (Koppe & Stotzek 1993, Boller & Häfliger 1996) and based on measurements.

Tab. 4.19 Transfer coefficients for metals in the WWTP process

	Transfer coefficient to raw sludge = share in particulate matter	Transfer coefficient to effluent = dissolved share
Aluminium	95%	5%
Arsenic	22%	78%
Cadmium	50%	50%
Chromium	50%	50%
Cobalt	50%	50%
Copper	75%	25%
Lead	90%	10%
Manganese	50%	50%
Mercury	70%	30%
Nickel	40%	60%
Silver	75%	25%
Tin	59%	41%
Zinc	70%	30%

Figures are not available for some elements. For these elements estimates are made based on their aqueous chemistry. Calcium and Magnesium are considered fairly soluble and the average particulate share is assumed to be 10%. The halogens (Cl, F, Br, I), potassium and sodium are assumed to be completely dissolved (particulate share 0%). Silver is assumed to behave like copper with a particulate share of 75%. For barium and silicon the same value as measured for aluminium is adopted (95%

particulate) based on the general low solubility of these elements. For all other elements a particulate share of 50% is assumed (B, Mo, Sb, Se, V, Be, Sc, Sr, Ti, Tl, W, Fe). For all estimated elements the uncertainty of the transfer coefficients to sludge and to effluent is increased, i.e. in Eq. 4.1 on page 27 the parameter *N* is doubled, e.g. a transfer coefficient of 50% will have a GSD of 104.8% instead of 102.4%.

Speciation of chromium

Chromium predominately exists in two oxidation states: a highly toxic, hexavalent form (Cr^{VI}) and the less toxic trivalent form (Cr^{III}). For the emissions of chromium the speciation must be decided. Hexavalent chromium forms oxianions (CrO_4^{2-}) which are soluble. In landfills it is observed that the chromium dissolved in landfill leachate is in fact Cr^{VI} (see part III on landfills). In aquatic environments, Cr^{III} readily sorbs onto organic particulates and co-precipitates with hydrous iron $FeO(OH)_x$ and manganese oxides MnO_y, while Cr^{VI} does not tend to sorb onto particulate matter to any significant extent (MacDonald 1994:58). Based on these informations it is fair to suggest that chromium emitted in the *effluent* of a WWTP is dissolved Cr^{VI}, while chromium retained in the *sludge* is adsorbed or precipitated Cr^{III}. Chromium in digested sludge disposed in landfarming is inventoried as Cr^{III} (cf. chapter 5.2 'Sludge to landfarming as fertiliser' on page 40). These are preliminary assignments and should be backed-up by measurements at WWTPs.

4.3.2 Uncertainties of emissions

In the inventory, the emissions are the product of wastewater composition and transfer coefficients. The uncertainty of the emissions figure depends on the uncertainty of those two factors.

Uncertainty of wastewater composition

The uncertainty of the wastewater composition expresses the variation of the average composition figures in the inventoried wastewater. The variation is dependent on the process producing the wastewater. The magnitude of pollutant concentrations is also influenced by the amount of water released as wastewater, e.g. if a substance is washed off with 1 litre of water or 10 litre of water will influence the concentration by a factor of 10. Unlike for solid wastes there is not necessarily a general correlation of uncertainty vs. concentration. The relative uncertainty of small concentrations can be similar to the relative uncertainty of large concentrations. Usually the uncertainties of wastewater concentrations are not known and must be estimated with the Pedigree approach.

Since the uncertainty of the wastewater composition depends on the process producing the wastewater, these uncertainties must be given along with the mean values of the wastewater composition. The calculation tool accommodates this input data.

Uncertainty of transfer coefficients

Transfer coefficients describe the fate of a substance or the frequency of a conversion path within the WWTP process. The uncertainty of the transfer coefficients can be understood as the *system-inherent* uncertainty of the WWTP that occurs when a hypothetical wastewater with *perfectly defined* composition is treated. This uncertainty can occur e.g. through stochastic processes in the plant. Conditions in the plant are heterogeneous in certain respects (solid/water boundaries) and can influence minor pathways of elements. Similar to the transfer coefficients in municipal waste incinerators (see part II) it can be argued that the smaller the transfer coefficient the larger the associated uncertainty is. I.e. large transfer coefficients denote a common and well established, robust pathway and minor variations will not contribute much to the relative uncertainty. On the other hand,

small transfer coefficients indicate uncommon pathways. Disturbances can influence these small pathways more than a major pathway with large transfer coefficient. The relative uncertainty of small transfer coefficients will therefore tend to be higher. However, the sensitivity towards disturbances in WWTP is diminished by longer residence times and better mixed media compared to waste incinerators.

To estimate the system-inherent uncertainty of the transfer coefficients, it is assumed here that the GSD for large transfer coefficients is close to 100%, i.e. major pathways of elements have a low uncertainty. For example particulate sulfur is completely transferred to the raw sludge and the uncertainty of that process is small, i.e. it is extremely unlikely that the average particulate sulfur to sludge transfer coefficient is 75% instead of 100%. On the other hand minor pathways are less well defined and the uncertainty is higher for small transfer coefficients. It is assumed that the GSD^2 is not above 150% even for minor pathways; i.e. there is a 95%-confidence range of the transfer coefficient value maximally between 67% and 150% of the mean transfer coefficient value, even for minor pathways[16].

A logarithmic formula to estimate the GSD of transfer coefficients (Eq. 4.1) is adopted from part II.

Eq. 4.1 $$GSD_{TK} = N \cdot \ln(m_{TK}) + c$$

where
N = –0.0346
m_{TK} mean value of transfer coefficient in [kg/kg]
c =1, (GSD_{TK} value for m_{TK} = 1)

The value of N is chosen so that a GSD^2 value of 150% is obtained for the smallest m_{TK} value (0.0017 kg/kg for nitrate to particulate nitrogen). For larger m_{TK} the GSD^2 value decreases and approaches 100% for m_{TK} = 1. The choice of N is based on the impression that the variation of the WWTP output depends mainly on the variability of the input wastewater. The variability of the treatment technology (which determines the variability of transfer coefficients) is not deemed to be very influential. It seems feasible that an average transfer coefficient of 10% could vary between 11.7% and 8.5%; or that an average transfer coefficient of 1% could vary between 0.74% and 1.34% (current model choice). However this data is not backed up by measurements. It is difficult to obtain such data as the variation the input *and* output would need to be determined for a large number of plants over a long time. The variability of transfer coefficients could then be determined by difference. The current estimate must suffice for the time being.

For the average operation conditions and with the current choice of transfer coefficients uncertainty, 87%-100% of the uncertainty in the WWTP output is determined by the *uncertainty of the input wastewater*. Only 0%-13%, depending on pollutant, is determined by the *uncertainty of transfer coefficients*. This reflects the impressions stated above.

[16] If the GSD is 122%, then GSD^2 is 150%. The 2.5%-confidence value is mean/GSD^2 = mean·67%. The 97.5%-confidence value is mean·GSD^2 = mean·150%. The Pedigree approach used to determine generic GSD^2 values within the Ecoinvent project suggests a basic uncertainty value of GSD^2 = 150% for emissions to water. The choices made here are compatible with the Pedigree approach.

4.4 Sludge digestion process

Digestion is assumed for all raw sludge. In new future plants, digestion might become less important in favour of sludge incineration. The choice depends on local conditions regarding cost of sludge incineration, cost of digestion & energy conversion, and cost of commercial energy consumption[17].

During sludge digestion, the raw sludge is altered. Constituents can be dissolved, transferred to digester gas or remain in the solid phase of the sludge. Dissolved matter returns to the wastewater input. Matter that is ultimately found in the effluent, is already heeded in the transfer coefficients of the wastewater purification. Here the conversion from sludge to digester gas shall be quantified.

Tab. 4.20 Composition of raw sludge and digested sludge derived from model calculation (dry mass)

	raw sludge g/m^3 sewage		**digested sludge** g/m^3 sewage	
Nitrogen	5.95	calculated N	2.34	– 60.3%
TOC	44.3	[2]	17.4	– 60.3%
PO4-P	2.09	calculated P	2.09	unchanged
Fe	4.93	in $FePO_4$	4.93	unchanged
Al	0.251	in $AlPO_4$	0.251	unchanged
S	2	calculated S	1.71	– 22.3%
Oxygen	59.9	remainder	52.5	– 12.8%
Hydrogen	5.53	C/H ratio = 1.25 [1]	2.18	C/H ratio = 1.25 [1]
sum	125		81.3	

1 from (von Raczeck 1993)

2 from TOC content in raw sewage(Tab. 4.4 on page 14)and transfer coefficient to sludge (Tab. 4.9 on page 17)

Total dry mass reduction during digestion is approximately 35% (BUWAL 1996:29), mainly achieved by conversion of carbon to methane CH_4 and carbon dioxide CO_2. The transfer coefficient for carbon to air is set, so that an overall reduction in sludge dry mass of 35% results. This transfer coefficient is 60.3%.

The transfer coefficient for carbon to air (60.3%) is also adopted for nitrogen, as the nitrogen in sludge is part of the biomass and converted to volatile N_2, N_2O and NH_3 under anaerobic conditions. All nitrogen in biogas must come from the sewage, as in anaerobic conditions no atmospheric N_2 is present. Not all this nitrogen in biogas is converted to nitrogen oxides in incineration (see nitrogen species profile in Tab. 4.21). No information could be found regarding the creation of thermal NO_x. Comparision with nitrogen flux in MSWI (see part II) suggests that all NO_x originates from fuel-N. All NO_x emissions were inventoried as waste-specific, i.e. if the wastewater does not contain nitrogen, no direct NO_x emissions will result in the WWTP inventory.During anaerobic digestion, also sulfur in the raw sludge can be converted to gaseous hydrogen sulfide H_2S and be transferred to digester gas. From the comparison of the sulfur in raw sludge (Tab. 4.20) and the sulfur found in digester gas (Tab. 4.21) a transfer coefficient for sulfur to gas of 22.3% is found.

Since the digester is anaerobic, all oxygen in the digester gas must originate from the sludge. Oxygen in the digester gas is essentially CO_2, i.e. the carbon transfer coefficient determines also the oxygen transfer coefficient. Carbon in raw digester gas is present as 40 mol% CO_2 (KfU 1992:107). The oxygen removed as CO_2 translates to a transfer coefficient of oxygen to gas of 12.8%.

[17] Personal communication with Max Maurer, EAWAG Dübendorf, March 5, 2002.

Tab. 4.21 Emissions and species profile from digester gas incineration

Source	Flue gas from incineration Zimmermann et al. 1996:C.27	Flue gas from incineration Column 1 divided by 13.74 Nm^3 gas / PCE*a [1]	Flue gas from incineration Column 2 multiplied by 0.068 Nm^3 gas /m^3 sewage [2]
unit	g/PCE*a	g/Nm^3 gas	g/m^3 sewage
CO	67	4.877	0.3317
CO_2	41500	3021	205.4
NMVOC	1	0.0728	0.00495
CH_4	220	16.02	1.089
SO_2	180	13.1	0.891
NO_2	135	9.828	0.6683
NH_3	15	1.092	0.07425
N_2O	10	0.728	0.0495
N_2		48.77 [3]	3.316

1 Divisor 13.74 Nm^3 gas / PCE*a obtained from 0.068 Nm^3 gas /m^3 sewage (BUWAL 1994:62) and 202 m^3/PCE*a (BUWAL 2001).

2 Factor 0.068 Nm^3 gas /m^3 sewage obtained from (BUWAL 1994:62), mean value for 88 WWTP in 1991.

3 Adjusted, so that NO_x emissions in model match the observed value of 0.6683 g NO_2/m^3 sewage. 91.8% of nitrogen in biogas is emitted as N_2 after incineration.

The emission of CO_2 is fully allocated to the carbon in digester gas. The emissions of CO, NMVOC and CH_4 are more dependent on incineration temperature, retention times, mixing etc. and in the MSWI incinerator were considered to be constant, process-specific emissions independent of the waste composition (see also part II on incineration). However, the generation of digester gas is mainly dependent on the carbon content in the wastewater. These emissions are therefore also allocated to the carbon in digester gas. The emission of SO_2 is allocated to sulfur in digester gas. The emissions of NO_2, NH_3, N_2O are allocated to the nitrogen in digester gas. Elemental nitrogen N_2 is nor inventoried in ecoinvent 2000, but is needed in the calculation for the nitrogen mass balance and species profile.

Volatile metals

Metals are usually believed to be non-volatile during the digestion process, due to the low temperatures <80°C involved. Measurements in low temperature environments of biologically active environments such as landfills and sewage treatment plants however have shown that some metals can become volatile in trace amounts by transformation to hydrides and organometals (metallocenes), especially methylated metals (Feldmann & Hirner 1995). Examples of compounds detected in such environments are listed in Tab. 4.22. Also selenium, tellurium, germanium, phosphorous, sulfur and silicon (Se, Te, Ge, P, S, Si) are known to from volatile compounds. In part III on landfills the volatility of metals in municipal solid landfills was already established, based on data from (Belevi & Baccini 1989).

Tab. 4.22 Some volatile (organo)metal compounds produced in biologically active environments.

Dimethyl mercury	$Hg(CH_4)_2$
Tetramethyl tin	$Sn(CH_4)_4$
Tin hydride	SnH_4
Trimethyl antimony	$Sb(CH_4)_3$
Trimethyl arsenic	$As(CH_4)_3$
Arsenic hydride	AsH_3
Trimethyl bismuth	$Bi(CH_4)_3$
Tetraethyl lead [1]	$Pb(CH_3CH_3)_4$

1 also produced in the technosphere as anti-knocking agent for motor fuels

For this study metal volatility for Cd, Pb, As, Hg, Sb, and Sn is estimated based on data given in (Feldmann & Hirner 1995) and reproduced in Tab. 4.23. These transfer coefficients are judged to be in the correct order of magnitude as compared with values from laboratory based sludge fermenters[18]. The significance of these air emissions is relative to the air emissions from sludge incineration, where also some trace metals are emitted; i.e. if these volatilisations are relevant or not in the inventory of air emissions depends on the share of sludge that is incinerated. In an average situation[19] the incineration is the most important source for metal air emissions for tin, lead and cadmium. For *mercury, arsenic and antimony* however the reverse is true: here the contribution from digester volatilisation is over 99% and is therefore the dominant source for air emissions in the total process chain WWTP→digester→sludge→incinerator. All these emissions are only considered, if the wastewater input contains the corresponding metals.

Tab. 4.23 Transfer coefficients from raw sludge to combusted digester gas

From raw sludge	Transfer coefficient to gas g/g	Comment	GSD
C	60.3%	Adjusted to result in correct overall mass reduction of 35%.	[3]
N	60.3%	Set equal to C, as N is in biomass.	[3]
S	22.3%	From SO_2 mass in gas vs. S mass in raw sludge	[3]
O	12.76%	From CO_2 to air	[3]
As	0.13%	Estimate based on data from (Feldmann & Hirner 1995)	117.2% [2]
Cd	0.000045%	Estimate based on data from (Feldmann & Hirner 1995)	115.0% [2]
Hg	0.00024%	Estimate based on data from (Feldmann & Hirner 1995)	121.3% [2]
Pb	0.0000037%	Estimate based on data from (Feldmann & Hirner 1995)	124.4% [2]
Sb	0.01%	[1]	220.9% [2]
Sn	0.000017%	Estimate based on data from (Feldmann & Hirner 1995)	124.8% [2]
other elements	0%	Set to zero	[3]

1 From a value of "<0.02% as trimethyl antimony " from a personal communication with Joerg Feldmann, Department of Chemistry, University of Aberdeen, UK, of 23 January, 2003. Sb can be expected to be similar to As (here with 0.13%).

2 Derived from measured ranges in (Feldmann & Hirner 1995)

3 See Eq. 4.1 on page 27.

[18] Personal communication with Joerg Feldmann, Department of Chemistry, University of Aberdeen, UK, of 23 January, 2003

[19] With a share of 53.1 w-% of sludge to incineration (class 3).

4.5 Energy balance

Electricity is used in a WWTP to aerate the activated sludge bed (stage 2), for the sludge digestion process, pumps, illumination etc. Heat is needed for heating the digester and general space heating. A part of these energy demands can be covered by the energy produced from incinerating digester gas. The remainder is covered from commercial sources.

In the inventory model the energy demand is allocated to the causing processes as far as possible. The gross energy demand is known from average operation. Specific energy consumption figures are derived from allocation of this average figure. The wastewater constituents then determine the *waste-specific gross energy demand*. Some of the energy is covered internally by energy from digester gas. How much digester gas is produced is another waste-specific figure depending on the carbon content of the wastewater. Heeding the waste-specific energy production the *waste-specific net energy demand* can be calculated.

Some figures are obtained from the water protection database of BUWAL, referring to data of 1994 (DBGS 1994). The new database (DBGS II) is currently in preparation at BUWAL within the project GEWISS (water information system) of Federal Office for Water and Geology FOWG and BUWAL (BUWAL 2002c).

4.5.1 Gross energy demand and allocation

The average grid electricity consumption of WWTP is 0.216 kWh/m^3 sewage (DBGS 1994). Additionally 0.043 kWh electricity per m^3 sewage are the gross production from the utilisation of digester gas (DBGS 1994). The gross electricity consumption is 0.28 kWh/m^3 sewage.

The average fuel oil consumption of WWTP is 0.14 MJ per m^3 sewage (BUWAL 1994:66)[20]. The average natural gas consumption is 0.187 MJ per m^3 sewage (DBGS 1994). Internally, 0.663 MJ heat per m^3 sewage are produced from digester gas (Kaufmann 2001, BUWAL 2001)[21]. The gross heat consumption is 0.99 MJ/m^3 sewage.

Energy consumption is distributed according to Tab. 4.24.

Tab. 4.24 Distribution of the energy demand in a WWTP (Müller 1994)

Process	Electricity	Heat (oil, gas)
Aeration in the activated sludge bed	70%	–
Sludge digestion	20%	90%
Miscellaneous (Pumps, space heat)	10%	10%

Electricity consumption for aeration is allocated to the oxygen uptake in the oxidation processes in the activated sludge bed (waste-specific), i.e. for NH_4, $N_{org.}$, or NO_2 to NO_3 and carbon to CO_2. Electricity consumption for sludge digestion is allocated to the total dry mass of raw sludge (waste-specific). Miscellaneous electricity consumption is allocated the wastewater input (m^3 sewage), i.e. is process-specific. Heat demand for digester heating is allocated to the total dry mass of raw sludge (waste-specific). Heat demand for space heat in the plant is allocated the wastewater input (m^3 sewage), i.e. is process-specific.

[20] From 920 tonnes of oil per year and 1.4 million PCE, converted using 202 m^3/PCE*a and 42.6 MJ/kg heating value.

[21] 956 TJ/a heat (Kaufmann 2001) from 1.44 Mio. m^3 sewage (BUWAL 2001). 100% internal consumption

4.5.2 Internal energy production and allocation

Energy is produced from the incineration of the digester gas. The average gross electricity produced is 0.043 kWh/m^3 sewage (DBGS 1994). The average gross heat produced is 0.663 kWh/m^3 sewage (Kaufmann 2001, BUWAL 2001). The main energy in digester gas is contained in methane CH_4. The energy production is therefore allocated to the carbon content in the digester gas. Per average m^3 sewage 16.4 g of C is transferred to the digester gas[22]. Per kg C in the digester gas 2.61 kWh electricity and 40.24 MJ useful heat are produced. These factors are used to calculate the *waste-specific energy production* depending on the wastewater composition, i.e. the carbon transferred to the digester gas. If no carbon is specified in the wastewater, no energy is produced.

4.5.3 Waste heat emissions

Waste heat is produced by the mineralisation of biomass, the incineration of digester gas and the net consumption of electricity. Per kilogram of C in direct emissions of CO_2 a waste heat production of 40.53 MJ is assumed (value for C in CH_2O from Michel 1938). For N in direct emissions of NO_2 a value of 6.276 MJ/kg N is assumed (Michel 1938). Waste heat from the activated sludge bed is assumed to be emitted to water. Waste heat from the sludge digestion process is assumed to be emitted 50% to water and 50% to air. Waste heat from digester gas incineration is assumed to be emitted 100% to air. The uncertainty of the waste heat emissions is calculated from the uncertainties in CO_2 and NO_2 emissions.

Within the ecoinvent 2000 project no waste heat emissions must be inventoried for electricity production processes. Accordingly, the waste heat generated in WWTPs must discount the energy contained in any net produced electricity. In the average case, all produced gross energy is consumed internally. If for a specific wastewater the net electricity production is larger than zero, the waste heat emissions to air are reduced accordingly.

4.6 Auxiliary material consumption

Auxiliary materials are used to precipitate phosphate ($FeCl_3$, $FeSO_4$, $Al_2(SO_4)_3$). Additional sulfate and chloride is found in the effluent, because the precipitating agents release it. The inventory of precipitating agents is explained in section 'Transfer coefficients for phosphorus' on page 19.

4.7 Process-specific burdens

Process-specific burdens are burdens in the WWTP which are not assigned to one specific component of the wastewater but to the whole treated wastewater for each m^3. A part of energy consumption is assigned process-specifically (see chapter 4.5.1 'Gross energy demand and allocation' on page 31).

Grit is solid waste that is removed from the wastewater before treatment. Grit consists of packaging, leafs and other waste that is typically washed from road surfaces to the sewers. So grit mainly stems from littering and road surfaces. It is debatable, if the disposal of this waste shall be assigned to the wastewater treatment in general, and not rather to the waste producer. However, it is a technical necessity to remove grit before wastewater treatment. In this sense grit disposal is assigned to the treatment[23].

[22] 67.3 g TOC /m^3 sewage and transfer coefficient to biogas 24.5%.

[23] This is similar to the removal and disposal of – mostly biomass – waste in rivers in river hydro electricity production. Here too it can be argued that the waste does not originate from the plant itself, but from other sources. Nevertheless, it is a →

An average figure of 31 g grit waste per m^3 wastewater is given in (UAG 1999). This amount is inventoried as 50% plastics mixture and 50% paper (instead of biomass) and disposed in a municipal waste incinerator. Grit is transported 10 km by lorry to the next waste incinerator.

Tab. 4.25 GSD^2 value for grit waste output from WWTP

Exchange	GSD^2 value	Pedigree scores	Comment
Grit waste output in WWTP	107%	(1,3,1,3,1,1)	Basic uncertainty of 1.05; data for WWTPs of Canton Aargau 1998 from (UAG 1999)

4.8 Infrastructure for wastewater treatment plant

Infrastructure modules in ecoinvent 2000 have the functional unit 'per plant' (i.e. unity). A fraction of this infrastructure module is inventoried per m^3 of sewage. This fraction is different for every class of WWTP, since the lifetime capacity per plant differs between WWTP classes. The capacity classes in PCE/a of Tab. 2.1 on page 2 can be converted to m^3/a using the factor of 202 m^2/PCE. Using a lifetime of 30 years per plant, the fraction of an infrastructure unit allocated to one m^3 of wastewater can be calculated.

Tab. 4.26 Calculation of infrastructure units per m^3 sewage

Class		1	2	3	4	5
Capacity average	PCE/a	233'225	71'133	24'864	5'321	806
Annual sewage volume in	m^3 /a	47'114'227	14'369'780	5'022'942	1'074'827	162'837
Lifetime plant	a	30	30	30	30	30
Lifetime sewage volume in	m^3	1'413'343'500	431'068'000	150'679'375	32'242'901	4'884'813
Plant infra per m^3 sewage	unit/m^3	7.075E-10	2.320E-09	6.637E-09	3.101E-08	2.047E-07

Infrastructure information of WWTP plants is available from two sources (Flückiger & Gubler 1994, Fahrner et al. 1995) covering three Swiss plants (Twann, Ergolz, Werhölzli). Both studies give inventories relating to one PCE, cf. Tab. A. 1 on page 50 in the appendix. For large plants the specific demands are lower than for the smaller plants. The infrastructure demands per PCE show good linearity versus the plant capacity. To calculate the average infrastructure data for the five different capacity classes, the linear relation established by the three data points is extrapolated to the average capacity of the three classes given in Tab. 4.26.

For some materials, Fahrner et al. (1995) has more detailed data for the plant Ergolz. To extrapolate this data to other capacity classes, the intercept of the linear regression with the x-axis, which is established for other materials, is adopted to these additional materials. The intercept for all metals, polymers, glass, mineral wool and chemicals is adopted from steel. The intercept for bitumen and lime is adopted from concrete.

Default ecoinvent transportation distances and modes are used for all materials. Steel, organic chemicals, bitumen, glass, copper, inorganic and organic chemicals are transported 600 km by train

technical necessity to remove that waste prior to water turbination, and it is prohibited to dump those wastes back into the river afterwards. Accordingly, the inventory of river hydropower should include the disposal of these wastes.

and 50 km by lorry. Aluminium, polyethylene, rubber, mineral wool and bitumen is transported 200 km by train and 50 km by lorry. Lime is transported 100 km by train and 20 km by lorry. Concrete is transported 20 km by lorry.

The sum of concrete and reinforcement steel (3.4% steel) is disposed as reinforced concrete in dismantling. The sum of glass and mineral wool is disposed as mineral wool in dismantling. The sum of polymers is disposed as polyethylene products in dismantling. All metals are assumed to be recycled.

4.8.1 Land use from plant infrastructure

Land use is inventoried in the infrastructure module. Area sizes for three plants are estimated from maps. Specific area sizes per PCE are calculated, cf. Tab. 4.27. The devised specific area sizes are linearly interpolated to the five capacity classes using the average capacity per class, cf. Tab. 4.28.

Tab. 4.27 Estimated area sizes for three WWTP plants

WWTP	Total area	Sealed area	PCE	Total area per PCE
	m^2			m^2
Liedertswil BL	300	50%	317	0.946
Wintersingen BL	1400 [1]	25%	630	2.222
Werdhölzli ZH	175'000	75%	593'300	0.295

1 including a rather large surrounding vegetation-covered area

Tab. 4.28 Land use figures for five WWTP classes

WWTP capacity class		1	2	3	4	5
Capacity average	PCE/a	233'225	71'133	24'864	5'321	806
Specific land use	total m^2/PCE	0.30 [1]	0.79	0.93	0.99	1.00 [2]
% sealed		75.0% [1]	57.6%	52.6%	50.5%	50.0% [2]
Total m^2 per plant	m^2	69'970	56'070	23'060	5'248	806.1
Sealed area per plant	m^2	52'480	32'270	12'130	2'650	403
Vegetated area per plant	m^2	17'490	23'790	10'930	2'599	403
Construction time	a	3	1.6	1.2	1.04	1

1 Estimated from WWTP Werdhölzli

2 Estimated from WWTP Liedertswil & WWTP Wintersingen

Plants are usually located near a river and are erected on formerly open, unsealed land. For all capacity classes the original land type 'pasture and meadow' (CORINE 231) is assumed (transformation from). Construction times are assumed to be 1 year for smaller plants and 3 years for larger plants; intermediate plants are linearly interpolated using the PCE figure (see Tab. 4.28). During the construction time the total area is inventoried as 'construction site' (CORINE 133). Land transformations to and from construction sites are not inventoried in ecoinvent 2000, but their land occupations are (Frischknecht et al. 2007). The final resulting plant area is inventoried as 'built-up industrial' (sealed) and 'vegetated industrial' (CORINE 121a and 121b). The plant operation time is assumed to be 30 years. After that time the plant is expanded, renovated or closed down.

Tab. 4.29 Life cycle inventory for 5 different capacity classes of municipal wastewater treatment plants.

Per unit (1 plant)	Location	Infrastructure	Unit	class 1	class 2	class 3	class 4	class 5
excavation, hydraulic digger	RER	0	m3	889'100	362'700	135'900	29'910	4'560
transport, lorry 28t	CH	0	tkm	12'640'000	5'137'000	1'923'000	423'200	64'510
transport, freight, rail	RER	0	tkm	14'950'000	6'076'000	2'275'000	500'600	76'310
electricity, medium voltage, at grid	CH	0	kWh	9'725	3'956	1'482	326	49.7
concrete, exacting, at plant	CH	0	m3	256'400	104'200	39'030	8'587	1'309
reinforcing steel, at plant	CH	0	kg	19'760'000	8'089'000	3'033'000	667'600	101'800
tap water, at user	RER	0	kg	31'220'000	12'710'000	4'763'000	1'048'000	159'800
aluminium, production mix, cast alloy, at plant	RER	0	kg	220'900	90'410	33'900	7'463	1'138
limestone, crushed, washed	CH	0	kg	5'501'000	2'236'000	837'300	184'200	28'080
chromium steel 18/8, at plant	RER	0	kg	1'593'000	649'100	243'200	53'510	8'159
glass fibre, at plant	RER	0	kg	498'100	203'900	76'450	16'830	2'566
copper, at plant	RER	0	kg	234'700	96'080	36'030	7'930	1'209
synthetic rubber, at plant	RER	0	kg	224'900	92'060	34'520	7'599	1'159
rock wool mat, packed, at plant	CH	0	kg	222'300	91'010	34'130	7'512	1'145
chemicals organic, at plant	GLO	0	kg	1'033'000	422'700	158'500	34'890	5'320
bitumen, at refinery	CH	0	kg	127'600	52'240	19'590	4'312	657
chemicals inorganic, at plant	GLO	0	kg	126'600	51'840	19'440	4'279	652
polyethylene, LDPE, granulate, at plant	RER	0	kg	4'093	1'676	628	138	21.1
polyethylene, HDPE, granulate, at plant	RER	0	kg	622'300	254'800	95'530	21'030	3'206
extrusion	CH	0	kg	626'400	256'400	96'160	21'170	3'227
disposal, building, reinforced concrete, to sorting plant	CH	0	kg	584'100'000	237'400'000	88'910'000	19'560'000	2'982'000
disposal, building, polyethylene/polypropylene products, to final disposal	CH	0	kg	851'300	348'500	130'700	28'760	4'386
disposal, building, bitumen sheet, to final disposal	CH	0	kg	127'600	52'240	19'590	4'312	657
disposal, building, mineral wool, to final disposal	CH	0	kg	720'400	294'900	110'600	24'340	3'711
Transformation, from pasture and meadow			m2	69'970	56'070	23'060	5'248	806
Occupation, construction site			m2a	69'970	56'070	23'060	5'248	806
Transformation, to industrial area, built up			m2	52'480	32'270	12'130	2'650	403
Transformation, to industrial area, vegetation			m2	17'490	23'790	10'930	2'599	403

Occupation, industrial area, built up	m2a	1'574'000	968'200	363'800	79'490	12'090
Occupation, industrial area, vegetation	m2a	524'800	713'800	328'000	77'960	12'090

4.8.2 Uncertainties for infrastructure

The Pedigree approach is used to estimate the uncertainty of the inventoried figures.

The uncertainty in energy consumption (electricity) is set equal to the uncertainty in material demand. This although the Pedrigree approach suggests smaller uncertainties for energy consumption (basic uncertainty of 105%). For energy demand in building activities it seems unlikely that the demand in materials has a basic uncertainty of 300%, but the energy used to process and handle those materials has only a basic uncertainty of 105%. Consequently, energy consumption for infrastructure has the same basic uncertainty as the materials (300%). The same reasoning applies to disposal services. If infrastructure materials with an input basic uncertainty of 300% are disposed, the basic uncertainty cannot change to 105% for the disposal of those materials (as suggested by the Pedigree approach for waste disposal processes).

Uncertainty of transportation services is calculated from the total transportation services for all materials and the Wilkinson-Fenton approximation for sums.

Tab. 4.30 GSD^2 values for material inputs and land use for WWTP infrastructure

Exchange	GSD^2 value	Pedigree scores	Comment
Infrastructure material	301%	(1,2,2,1,1,4)	Basic uncertainty of 3; data from 3 plants
Land transformations	202%	(2,3,1,1,1,4)	Basic uncertainty of 2; estimate on land area from 3 plants
Land occupations	153%	(2,3,1,1,1,4)	Basic uncertainty of 1.5; estimate on land area from 3 plants, class capacity size and building, operation time

5 Sludge disposal

Three options exist for the disposal of the digested sludge: incineration, recycling as fertiliser or landfilling (chapter 3.5 'Disposal of WWT sludge' on page 7).

Tab. 5.1 Average disposal of digested WWTP sludge in Switzerland (w% dry substance)

	Source	to agriculture	to landfill	to incineration
1994 average	Zimmermann et al. 1996:C.45 , from BUWAL WWTP database	49.0%	19.0%	32.0%
2000 average	(BUWAL 2001a)	38.6%	2.4%	59.0%
Trend	(Stadelmann et al. 2002)	Decreasing	Decreasing	Increasing
Comment	(BUWAL 2002a)	Disallowed in 2003-2005	Disallowed in 2000	Goal sink

In (Zimmermann et al. 1996:C.45) information on disposal types *for different WWTP classes* are given for 1994. In large WWTPs (Class 1) the sludge was predominantly incinerated or landfilled; in small WWTPs (class 5) the sludge was rather recycled as fertiliser. Since a large part of the Swiss wastewater is treated in large WWTP, most sludge is disposed as waste. The data from 1994 is extrapolated to 2000 with the following assumptions:

1. Neglecting a minor share of 2.4% mandated exceptions, landfilling of sludge is not legal in Switzerland anymore and is not considered a viable disposal path for any WWTP class in 2000.
2. The class-specific share to agriculture of 1994 is decreased by a factor 0.744. This leads to a weighted average share to agriculture of all plants that equals the average share in 2000 of 38.6% (cf. Tab. 5.1).
3. The remainder of the sludge is incinerated.

The extrapolated figures in Tab. 5.2 are used as default disposal paths for digested sludge. The user has the possibility to override those figures in the calculation tool.

Tab. 5.2 Estimated disposal paths of digested WWTP sludge for different WWTP capacity classes in 2000 (w% dry substance)

WWTP capacity Class		to agriculture	to landfill	to incineration
Class 1	Extrapolation of 1994 data	28.26%	0%	71.7%
Class 2	Extrapolation of 1994 data	38.67%	0%	61.3%
Class 3	Extrapolation of 1994 data	46.86%	0%	53.1%
Class 4	Extrapolation of 1994 data	71.40%	0%	28.6%
Class 5	Extrapolation of 1994 data	70.65%	0%	29.3%
Weighted average		38.6%	0%	61.4%
Water content, w%	Zimmermann et al. 1996:C.45, from BUWAL WWTP database	93.3%	92.0%	63.0%

The uncertainty of the disposal path shares is estimated with a logarithmic formula (Eq. 5.1). If the share is close to 100% the uncertainty is assumed to be low, resulting in a GSD value of near 100%. Smaller shares have larger uncertainty. These uncertainties are not independent, which is however not heeded here. The uncertainty is assumed to cover the uncertainty introduced by extrapolation of 1994 data. For the smallest disposal path shares in Tab. 5.2 only moderate GSD values of 106.3% result.

Eq. 5.1 $$GSD_S = N \cdot \ln(S) + 1$$

where

GSD_S Geometric standard deviation (uncertainty) of the disposal path share *S*

N = –0.05

S mean value of the disposal path share

5.1 Sludge incineration

Prior to waste incineration the sludge is transported 10 km by lorry to the waste incinerator plant. A water content of 63% is included in this transport.

The inventory of the incineration of digested sludge can be calculated with the model developed in part II (MSWI incineration) and part III (landfill for incineration ashes)[24]. The following procedure is applied to calculate the waste-specific burdens from incineration.

An *average* sludge composition is defined as waste to incineration[25]. The average WWTP sludge composition is shown in Tab. 5.3. The calculated *average* burdens (emissions, material consumption etc.) in the MSWI from this sludge are allocated to the causing masses in the Average sludge. From the relation of causing masses and resulting MSWI burdens, factors are calculated for the *burden per mass unit of a chemical element* ('burden factors'). For *waste-specific* sludges from wastewater treatment, the resulting sludge composition from the WWTP model are then multiplied with these burden factors, resulting in *waste-specific* burdens from sludge incineration. The allocation key of MSWI burdens to the sludge components is shown in Tab. 5.4.

Tab. 5.3 Average WWTP sludge composition to incineration.

kg/kg		kg/kg		kg/kg	
H_2O	0.63	Ba	0	Zn	0.00057
O	0.00124	Cd	0.00000104	Be	0
H	0.0163	Co	0.00000601	Sc	0
C	0.13	Cr	0.0000455	Sr	0
S	0.0116	Cu	0.000209	Ti	0
N	0.0176	Hg	0.00000104	Tl	0
P	0.0164	Mn	0.000197	W	0
B	0	Mo	0.00000356	Si	0.0222
Cl	0	Ni	0.0000196	Fe	0.1
Br	0	Pb	0.000058	Ca	0.0376
F	0	Sb	0	Al	0.0111
I	0	Se	0	K	0
Ag	0	Sn	0.0000149	Mg	0.00422
As	0.00000147	V	0	Na	0

0 For values of zero a very small dummy value (1.23E-9) is used to be able to calculate burden factors also for elements that have no established concentration in average WWTP sludge.

[24] These two models are integrated in the Excel tool 13_MSWI.xls.

[25] The average WWTP sludge composition is obtained from the WWTP model developed here and an average Swiss wastewater composition as an input to WWTP. The average Swiss wastewater composition is detailed in part I (waste compositions).

Tab. 5.4 Allocation key of MSWI burdens to causing masses in digested WWTP sludge

	Carbon dioxide, biogenic to air and BOD5, COD, TOC, DOC, to water	C
	Sulfur dioxide to air and sulfate to water	S
	Nitrogen oxides, ammonia, dinitrogen monoxide, cyanide, to air and nitrate to water	N
	Phosphorus to air and phosphate to water	P
	Boron to air and boron to water	B
	Hydrogen chloride to air and chloride to water	Cl
	Bromine to air and bromine to water	Br
	Hydrogen fluoride to air and fluoride to water	F
	Iodine to air and to water	I
	Silver to air and to water	Ag
	Arsenic to air and to water	As
	Barium to air and to water	Ba
	Cadmium to air and to water	Cd
	Cobalt to air and to water	Co
	Chromium to air and to water	Cr
	Copper to air and to water	Cu
	Mercury to air and to water	Hg
	Manganese to air and to water	Mn
	Molybdenum to air and to water	Mo
	Nickel to air and to water	Ni
	Lead to air and to water	Pb
	Antimony to air and to water	Sb
	Selenium to air and to water	Se
	Tin to air and to water	Sn
	Vanadium to air and to water	V
	Zinc to air and to water	Zn
	Beryllium to air and to water	Be
	Scandium to air and to water	Sc
	Strontium to air and to water	Sr
	Titanium to air and to water	Ti
	Thallium to air and to water	Tl
	Tungsten to air and to water	W
	Silicon to air and to water	Si
	Iron to air and to water	Fe
	Calcium to air and to water	Ca
	Aluminium to air and to water	Al
	Potassium to air and to water	K
	Magnesium to air and to water	Mg
	Sodium to air and to water	Na
	municipal waste incineration plant	waste mass 1
	process-specific burdens, municipal waste incineration	waste mass 1
	Carbon monoxide, biogenic	waste mass 1
	Methane, biogenic	waste mass 1
	slag compartment	slag mass 2
	process-specific burdens, slag compartment	slag mass 2
	residual material landfill facility	residue mass 3
	process-specific burdens, residual material landfill	residue mass 3
	Heat, waste	C
	Heat, waste	C
	electricity from waste, at municipal waste incineration plant	waste mass 1
	heat from waste, at municipal waste incineration plant	waste mass 1
	sodium hydroxide, 50% in H2O, at plant	S N P Cl Br F I in scrubber
	quicklime, milled, packed, at plant	N P Cl Br F I in scrubber
	hydrochloric acid, 30% in H2O, at plant	mass in scrubber
	iron (III) chloride, 40% in H2O, at plant	heavy metals in scrubber
	chemicals organic, at plant	Hg, Cd in scrubber
	chemicals inorganic, at plant	mass in scrubber
	cement, unspecified, at plant	residue mass 3
	cement to residual landfill	residue mass 3
	ammonia, NH3, liquid, at regional storehouse	N in scrubber
	natural gas, burned in industrial furnace low-NOx >100kW	N in scrubber
	titanium dioxide, at plant	N in scrubber
	chromium oxide, at plant	N in scrubber

1 Waste mass is the total waste input sum *without water, oxygen and hydrogen*. The mass of water, oxygen and hydrogen is allocated to the other elements in proportion to mass.

2 Slag mass includes the oxygen in oxides.

3 Residue mass includes the oxygen in oxides.

The necessary transport services are calculated from the waste-specific masses and standard distances as used in the MSWI model (see part II).

The calculated burden factors constitute a sparsely filled matrix of 37 columns (allocand elements) by 155 rows (allocated MSWI unit process inventory exchanges). A representation in print would take up over 14 pages. The full matrix is contained in the calculation tool in the sheet 'sludge incineration' of the Excel workbook '13_WWT.xls' on the CD-ROM.

5.1.1 Uncertainty of incineration burdens

The inventory data calculated in the MSWI model includes uncertainty ranges. These ranges have also to be adjusted to match the *waste-specific* ranges in the input sludge obtained in the WWTP process.

First, for the calculation of the 'burden factors' the uncertainty of the sludge input to MSWI is set to the minimum (all $GSD^2 = 100\%$)[26]. This way the uncertainty of the MSWI inventory data shows only the uncertainty introduced *by the MSWI process alone*, and not the combined uncertainty of the waste input plus MSWI process. The resulting uncertainty in the MSWI inventory is used for the uncertainty of the burden factor alone.

Next, the uncertainty in the *allocand masses* are calculated. For single chemical elements this is the GSD value calculated of the corresponding component in the digested sludge. For sums the Wilkinson-Fenton approximation is used to calculate a GSD value.

The waste-specific burdens from sludge incineration are obtained by multiplying the amounts of allocands in the waste-specific composition of the digested sludge with the 'burden factors'. Accordingly, the GSD value for these burdens is obtained by using the uncertainty propagation formula for multiplication with the GSD values of the factors in the product.

5.2 Sludge to landfarming as fertiliser

Landfarming is the use of sludge as fertiliser in agriculture. Swiss regulations limits wastewater sludge application to 5 tons of dry substance per hectare over 3 years[27] (Annex 4.5 in StoV 1999).

5.2.1 Disposal or recycling?

Sludge contains plants nutrients such as phosphorus, nitrogen, potassium etc. which might be considered beneficial in agriculture. Application of wastewater treatment sludge is especially important for closed phosphorus cycling (Herter et al. 2001). This raises an allocation question, if the burdens from sludge application are to be allocated to the wastewater producer or the agricultural production. Within the ecoinvent 2000 project a cut-off boundary solution is prescribed for recycled or downcycled materials produced from a process, while disposed waste materials are usually accounted in the waste-producing process (Frischknecht et al. 2007). No gradual allocation is performed for recycled materials. The question is then if sludge can be regarded as a valuable commodity or as a (negative value) waste material.

[26] This can be achieved by setting the factor N in the generic uncertainty calculation of the waste composition (i.e. GSD=N*ln(m)+1) to zero. See part II.

[27] With a water content of 93% in sludge, this equates to 2.4 kilograms of wet sludge per m^2 and year.

Municipal wastewater sludge is used by Swiss farmers as a zero value commodity, i.e. farmers do not pay to receive sludge, neither obtain a fee (Zimmermann et al. 1996:C.48). In that respect, sludge cannot be seen as a 'valuable' fertiliser. As pointed out in chapter 3.5 'Disposal of WWT sludge' on page 7, landfarming of sludge is decreasing due to saturated farm nutrient budgets, but also concerns about pollutants in the sludge. In a multi-criteria risk-benefit analysis for different materials, wastewater treatment sludge received the lowest ranking of all considered materials[28] (Herter et al. 2001:250). Agricultural application of wastewater treatment sludge is being completely phased out in Switzerland by 2005. This questions the beneficial fertiliser properties of sludge. Landfarming of sludge rather has the character of a cheap disposal route for a problematic waste. It makes sense to account the final disposal of sludge to the wastewater producer and not the agricultural management.

On the other hand in the inventory of agricultural processes, the practice of *not* accounting emissions from wastewater sludge (because they are attributed to the wastewater producer) would cause astonishment: While the emissions e.g. of mineral fertilisers and farm dung are fully attributed to the agricultural production, the emissions from sludge would not. This would lead to questionable conclusions from LCA, that fertilising with sludge is superior than other fertilisers. This view is inappropriate. LCAs for farming are obviously interested to account the emissions occurring on the fields which are caused by farm management practice. Also the 'zero price'-argument used above can be also be reversed to the effect, that the sludge is not a waste, because if it were, the WWTP would have to pay for its disposal[29].

There are good reasons to include the sludge application as a waste in the LCI of wastewater treatment, but equally justifiable reasons can be found to set the cut-off boundary to include sludge application in agriculture as a recycled material. A coherent choice must be made within the ecoinvent database[30]. The problem is currently alleviated by the fact that wastewater sludge application is indeed *not* a considered process in the ecoinvent 2000 inventories of agricultural production (Nemecek et al. 2007).

In this study sludge application is attributed completely to the wastewater treatment[31]

This choice is mainly based on the fact that sludge application will be completely prohibited by October 2005, thus emphasising the waste characteristics of sludge.

[28] Other materials were farm dung, mineral fertilisers, compost, wood ash and two industrial production sludges. Wastewater treatment sludge only ranked highest in the criteria 'saved cost' and 'phosphor/potassium scarcity' but ranked last in the criteria 'organic pollutants', 'pathogens', 'BSE risk', 'dissipation of genetically altered organisms', and 'market/image risks'. Other of the weighted criteria in the risk-benefit analysis were 'perseverance of soil structure' and 'heavy metals'.

[29] However, price not always just reflects the intrinsic value of a product (waste or by-product), but also the competition involved or lack thereof. For example, waste tyres are burnt in cement kilns and the cement kiln operators receive a fee for doing so. This positive disposal price would categorise old tyres as a waste and not a valuable commodity. But to the cement kiln, waste tyres actually represent a value, since it allows them to (partially) cut down fuel input. And in ecoinvent 2000 waste tyres are actually inventoried as alternative fuel and fully attributed to the cement production for these reasons (Kellenberger et al. 2007). The positive price for accepting waste tyres therefore not only expresses the intrinsic value of the actual material, but also expresses the kiln operators awareness that competing channels that accept unusable tyres would charge for doing so. In this light the zero-price of wastewater treatment sludge can be seen as a trade-off between the positive fertiliser value of sludge and the negative value of the unwanted components. Farmers might well become aware of their unique position of accepting sludge and insist on a positive net fee for them to accept sludge in the future.

[30] Using *both* system boundary choices can lead to potentially inconsistent inventories, e.g. in potato starch production, if the sludge application would be included in the potato farming, but *also* included in the treatment of starch production wastewater.

[31] In accordance with Thomas Nemecek of FAL, personal communication of December 19, 2002.

Other possible allocation methods

The cut-off choice made above shall not prejudice other possible solutions. Instead of cut-off boundaries, as prescribed by the ecoinvent methodology, other, non-100% allocation factors could be found. Allocation could be based, for example, on costs for the fertiliser function and the waste disposal function of sludge application. The real costs of sludge are zero, implying that the costs of the individual functions are approximately equal[32]. This would suggest a 50%/50%-allocation of the burdens from sludge application between wastewater treatment and agriculture. Alternatively 'avoided costs' derived from costs for sludge incineration and the costs of equivalent amounts of nutrients from mineral fertiliser, could be used for allocation. A helpful approach is presented in (Frischknecht 2000). A 2-dimensional scheme is developed which helps to describe the above allocation dilemma. The scheme enables to identify allocation factors for which environmentally – if possible – favourable decisions are made *by both actors* confronted with sludge. The result depends on the burdens of alternative services fulfilling the same function.

It is not possible, nor prescribed, to perform such allocation analyses in this study. The cut-off boundary is used here as described above. It might be replaced by other methods in the future.

5.2.2 Technosphere exchanges

Process burdens encompass the burdens for transport and the spreading. Sludge to landfarming has a high water content of 93.3%, which is processed as well (Tab. 5.2). The burden to transport and spread the water is allocated to the dry substance contents of the sludge. For every kilogram of dry sludge 14.9 kg of water are processed in the wet sludge. Transport distance from the WWTP to landfarming is assumed to be 20 km by lorry[33] (estimate from Zimmermann et al. 1996:C.46).

As a proxy for distribution of the digester sludge on the field, the module 'slurry spreading, by vacuum tanker' is used[34]. This module is intended for application of farm manure slurry and has the functional unit 'm^3'. To convert the digester sludge calculated in kilogram to m^3 an estimated density of 1.03 kg/m^3 is used.

5.2.3 Biosphere exchanges

Accounting emissions to soil

Sludge is applied directly on agricultural surfaces. The ecoinvent database discerns such emissions as 'to soil, agricultural'. The fate of soil emissions is usually – if at all – heeded in the LCIA method, for example in CML'01 (Guinée et al. 2001). In LCIA *generic average fate factors* for a large reference region, e.g. Western Europe, are applied, since information on the geographical location of an emission is usually not available (anymore) at that stage[35]. Here the application of sludge to agricultural areas is conceptually treated like application of fertiliser, i.e. *local fate factors* which are specific to sludge are applied. Similar local fate calculations are performed for the application of farm dung or mineral fertilisers within the LCI of agricultural production in the ecoinvent 2000 project (Nemecek et al. 2007). For reasons of database consistency similar factors are also used for the

[32] I.e. the WWTP is willing to pay the farmer as much for sludge disposal as the farmer is willing to pay to receive sludge as fertiliser, leading to a net price of zero.

[33] Data module: transport, lorry 28t | CH | transport systems | road | 0 | tkm

[34] Data module: slurry spreading, by vacuum tanker | CH | agricultural means of production | work processes | 0 | m3

[35] Also in the ecoinvent database, a geographically differentiated LCIA of emissions is not possible yet, although processes are recorded along with their geographical location (country code).

application of WWTP sludge[36]. If specific local fate factors shall be introduced into the data, it makes sense to do so directly at the inventory stage, where the location of the emission is explicit, and not at the impact assessment stage, where emissions from all regions are cumulated.

Local fate factors for nitrogen: 43.4% of the nitrogen in sludge is present as NH_3 (Külling et al. 2002). 60% of the NH_3 is emitted to air (Menzi et al. 1997). A part of the emitted NH_3 is subsequently transformed to nitrous oxide in the atmosphere or after deposition (Schmid et al. 2000). This is the indirect N_2O. Usually LCIA methods do not consider this transformation of ammonia to N_2O. Nitrous oxide is a strong greenhouse gas with a GWP of 270 CO_2-equivalents, but LCIA methods give no global warming impact for ammonia emissions. Therefore, creation of indirect N_2O is heeded in the local fate factors. Of the remaining nitrogen in soil, 1.25% are transformed to N_2O in microbial denitrification processes and also emitted to air (Schmid et al. 2000). This is the direct N_2O. In all 27% of the nitrogen in sludge is lost to the atmosphere, 25.81% as NH_3-N and 1.18% as N_2O-N. This figure is highly variable depending on weather conditions. A range of 10% to 90% ammonia losses is given in (Herter et al. 2001:263). From that range an uncertainty value of 173% (GSD) is estimated for all nitrogen fate factors.

All remaining nitrogen is taken up by the plant and removed with the crop. The fate of this nitrogen is not further heeded (cut-off boundary). A small amount of elemental nitrogen emission (N_2) is also possible from microbial denitrification processes, but not quantified here. No accumulation of nitrogen in soil is assumed, and accordingly no emission to soil, ground- or surface water. It is assumed that the sludge is applied at an appropriate time and that no excess nitrate is washed off from the field. Nitrate emissions would be high for inappropriate sludge applications during winter.

Tab. 5.5 Local fate transfer coefficients for nitrogen for agricultural sludge application.

	kg/kg N	GSD
N to air as NH_3-N (net amount)	25.80%	173.2%
N to air as N_2O-N (direct + indirect)	1.18%	173.2%
N to water as NO_3-N	0.0%	–
N to plant or to air as N_2 #	73% #	–

not considered here

Local fate factors for phosphorus: A minor amount of phosphorus is washed off the field by intense precipitation events and by erosion. The transfer coefficient for phosphorus from sludge to groundwater is 0.57%; the transfer coefficient to surface water is 2.005%[37]. All remaining phosphorus is assumed to be taken up by the plant and removed with the crop. No accumulation of phosphorus in soil is assumed. Again, the fate of phosphorus in the plant is not further heeded (cut-off boundary).

[36] Attention must be paid to avoid a possible double consideration of fate in the LCI *and* LCIA, depending on the LCIA method. The soil emissions must be divided into plant uptake and emissions to water and air, *but not to soil*, as such an emission category would in some LCIA methods *again* be subjected to a fate calculation.

[37] Both factors from a personal communication with Volker Prasuhn, FAL Swiss federal research station for agroecology and agriculture, Reckenholz, Zürich, August 28, 2002.

Tab. 5.6 Local fate transfer coefficients for phosphorus for agricultural sludge application.

	kg/kg P	GSD
P to river water as PO_4-P	2.005%	124.9%
P to groundwater as PO_4-P	0.57%	124.9%
P to plant #	remainder #	–

not considered here

Tab. 5.7 GSD^2 value for phosphorus fate factors

Exchange	GSD^2 value	Pedigree scores	Comment
P fate factors	156%	(1,2,1,1,n.a.,5)	Basic uncertainty of 1.5; data from Volker Prashun, FAL.

Local fate factors for other elements: For other elements no local fate is considered in the inventory and they are inventoried as 'emission to agricultural soil'. Fate considerations, like plant uptake, are heeded – if at all – in the LCIA method. Uncertainty of emissions is calculated from initial wastewater uncertainty and uncertainty of transfer coefficients.

Burden or benefit from nutrients?

Nutrients can be considered a benefit on an agricultural area, if the nutrient input to the area is not in excess. The potential damage of nutrients is eutrophication. Eutrophication occurs if the nutrients input flux onto a certain area is above a tolerable threshold. This threshold is strongly dependent on local conditions. Accordingly the potential damage of nutrients releases is dependent on the region the release is occurring. This has lead to calculation of nation-specific LCIA characterisation factors for eutrophication damages, e.g. Huijbregts (1999) calculates eutrophication characterisation factors of air emissions for different European regions. Such regionally resolved factors are also possible for soil emissions. If nutrients are released on an sub-critically loaded area, less damage occurs compared to an area that is already nutrified in excess.

In this study, local fate for nitrogen and phosphorus is already included in the inventory (cf. above). Average local fate and plant uptake rates for Switzerland are applied, which diminish the eutrophication potential of a soil emission. In that sense, the anticipated fertiliser benefit from sludge alleviates some of the full eutrophication potential present in sludge[38]. Pollutants with no beneficial value to the plant are fully accounted as soil emissions and can lead to damages to humans via the ingestion of food from crops.

5.3 Unit Process Inventories

The inventory for the WWTP and the sludge disposal are integrated into one dataset per waste. The combined unit process inventories (i.e. non-cumulated data) of all wastewater compositions to WWTP represent a very large table of 90 columns and 240 rows. A table representation in print is not very sensible, as many cells contain no data and some comment cells contain long text entries. The inventoried data is therefore contained in the Excel table '13_WWTP_LCI.xls' on the CD-ROM. This workbook contains the list of exchanges in ECOSPOLD format.

[38] In an LCIA method heeding plant uptake of nutrients as a fate pathway, the same damage-diminishing effect is achieved. An LCIA method will, however, usually apply average generic fate factors referring to a larger region, e.g. Western Europe.

6 Calculation manual

Due to changes in the nomenclature of exchanges with the nature (resources, emissions to air, emissions to water) as well as in the nomenclature of datasets the here described spreadsheet calculation tools for the calculation of own waste datasets are not working correctly anymore. An update and adaption of these various calculation tools is foreseen within the work for version v2.1 – until then, these spreadsheet calculation tools are not available to the user of ecoinvent!

Together with this report calculation tools are distributed which allow the calculation of wastewater treatment inventories of user-specified wastewaters. The calculation tools are Excel spreadsheets and run on Microsoft Excel software. In the following a step-by-step guide to calculation of inventories of your wastes.

1. open the Excel file '13_WTTP.xls'.
2. Do not update links to other sheets. There is no need. Choose 'No' in the dialog.
3. On slower machines you might want to disable automatic calculation. Choose the menu command 'Extras/Preferences', choose 'calculation' and set the calculation to 'manual'.
4. In '13_WTTP.xls' go to the sheet 'input'. In column H you can enter the composition data for your wastewater (mean values). Please note that the unit is 'kilograms per cubic meter wastewater' and all entries relate to chemical elements, *not compunds*, e.g. ammonia (NH_4) is entered as nitrogen in ammonia (NH_4-N). Please note the valid combinations of parameters mentioned in column D.
5. In column I you can enter the uncertainty of your mean values as geometric standard deviation (GSD, lognormal distribution). Please note that GSD must be entered, not GSD-squared.
6. Below the composition data in cell H64 you must specify the desired capacity class of the WWTP (number 1 to 5). Large plants have small numbers. Swiss average and default value is capacity class 3. See Tab. 2.1 on page 2 for further reference. This entry will be used in creation of the module name.
7. In cell H65 and H66 you can enter the distribution of digested sludge to disposal. Options are incineration and application in agriculture. If no values are entered the default values for the specified capacity class are used (see Tab. 5.2 on page 37).
8. In cell H67 you can enter if the inventory shall contain infrastructure for residential sewage collection (see chapter 4.1 'Sewer system' on page 10).
9. You have finished defining the wastewater. Please enter now an English and German name for the wastewater at the top in cells H8 and H9. These entries will be used in creation of the module name. The names describe the wastewater and *not the complete module name*, i.e. the English entry NAMEXY will show up in the module name as 'treatment, NAMEXY, to wastewater treatment, class N' (where N is the specified WWTP capacity class).
10. To inventorise several wastewater compositions, for convenience your finished data set can be copy/pasted to the 'data base area' located on the right side starting from column AA.
11. Save the sheet.
12. Force Excel to recalculate the workbook by pressing 'Alt – =' in Windows machines or 'Command – =' in Mac machines or by choosing the menu command 'Extras/Preferences', 'calculation' and clicking on 'Calculate now'. Important: wait until calculation is complete. Excel's status bar comment in the lower left corner of your screen shows you the progress of the calculation.
13. Go to the sheets 'X-Process' and 'X-Exchange' from where you can copy the inventory data of the treatment of your wastewaste in a WWTP in ECOSPOLD format. Paste the tables *as values* to another sheet (press 'Alt–Shift–V' resp. 'Command–Shift–V' and choose 'values').

7 Cumulative Results and Interpretation

7.1 Introduction

Selected LCI results and values for the cumulative energy demand are presented and discussed in this chapter. Please note that only a small part of the about 1000 elementary flows is presented here. The selection of the elementary flows shown in the tables is not based on their environmental relevance. It rather allows to show by examples the contributions of the different life cycle phases, or specific inputs from the technosphere to the selected elementary flows. Please refer to the ecoinvent database for the complete LCIs.

The shown selection is not suitable for a life cycle assessment of the analysed processes and products. Please use the data from the database for your own calculations, also because of possible minor deviations between the presented results and the database due to corrections and changes in background data used as inputs in the dataset of interest.

The ecoinvent database also contains life cycle impact assessment results. Assumptions and interpretations were necessary to match current LCIA methods with the ecoinvent inventory results. They are described in (Frischknecht et al. 2003c). It is strongly advised to read the respective chapters of the implementation report before applying LCIA results.

7.2 Results for wastewater treatment

The Tab. 7.1 shows some arbitrary results of the cumulated inventory of wastewater treatment. Four wastes were chosen:

- Average municipal sewage to WWTP class 1 (large scale)
- Average municipal sewage to WWTP class 3 (medium scale)
- Average municipal sewage to WWTP class 5 (small scale)
- Unpolluted water to WWTP class 3 (medium scale)

The results refer to the treatment of those wastewaters and all further downstream burdens.

Tab. 7.1 Selected LCI results and the cumulative energy demand for municipal wastewater treatment

		Name		treatment, sewage, to wastewater treatment, clas 1	treatment, sewage, to wastewater treatment, clas 3	treatment, sewage, to wastewater treatment, clas 5	treatment, sewage, unpolluted, to wastewater treatment, clas 3
		Location		CH	CH	CH	CH
		Unit		m3	m3	m3	m3
		Infrastructure		0	0	0	0
LCIA results			Unit				
	cumulative energy demand	non-renewable energy resources, fossil	MJ-Eq	2.6400	3.5900	4.4800	2.9700
	cumulative energy demand	non-renewable energy resources, nuclear	MJ-Eq	1.5500	1.7100	1.8600	0.6170
	cumulative energy demand	renewable energy resources, water	MJ-Eq	0.5300	0.5790	0.6270	0.2010
	cumulative energy demand	renewable energy resources, wind, solar, geothermal	MJ-Eq	0.0312	0.0372	0.0435	0.0189
	cumulative energy demand	renewable energy resources, biomass	MJ-Eq	0.0802	0.1040	0.1280	0.0718
LCI results			Unit				
resource	Land occupation	total	m2a	0.00937	0.0149	0.0176	0.0125
air	Carbon dioxide, fossil	total	kg	0.238	0.316	0.384	0.278
air	NMVOC	total	kg	0.000177	0.000244	0.000301	0.000216
air	Nitrogen oxides	total	kg	0.00157	0.00189	0.00215	0.00105
air	Sulphur dioxide	total	kg	0.0013	0.00144	0.00157	0.00043
air	Particulates, < 2.5 um	total	kg	0.000203	0.000301	0.000386	0.000285
water	BOD	total	kg	0.0105	0.0106	0.0108	0.000707
soil	Cadmium	total	kg	3.93E-08	6.51E-08	0.000000098	1.81E-10
Further LCI results			Unit				
water	Cadmium, ion	total	kg	0.000000312	0.000000361	0.000000376	0.000000175
water	Copper, ion	total	kg	0.0000323	0.0000289	0.0000241	0.00000673
water	Lead	total	kg	0.00000657	0.00000831	0.00000846	0.00000671
water	Zinc, ion	total	kg	0.0000995	0.000123	0.000144	0.0000716

The first three datasets describe the treatment of identical wastewater in plants of different sizes. Of these three, the largest plant (class 1) has the lowest burdens, while the smallest plant (class 5) has the highest burden. The reason for this is the economy of scale of the infrastructure of large plants. A large plant makes more efficient use of the sewer and plant infrastructure. For a small scale plant comparatively large infrastructure expenditures must be invested for a relatively small amount of sewage input.

The copper emissions are the only exchange in Tab. 7.1 where the small scale plant has the lower burden than a large scale plant. The reason for this is that copper emissions are dominated by waste-specific emissions, i.e. depend strongly on the wastewater composition, while for other metals indirect burdens from infrastructure and energy consumption are more important. In a small scale plant more digester sludge is disposed on agricultural areas, (landfarming) and less in incineration. Sludge incineration will transfer copper to landfills and partly to ground water, while landfarming of sludge is inventoried as emission to soil. In a small scale plant ultimately less *water* emissions of copper are produced, but the emissions of copper to *soil* will be compensatorily larger.

The last column in Tab. 7.1 shows the 'treatment' of unpolluted water in a class 3 WWTP. I.e. here are no pollutants to treat, no sewage sludge will be generated and no pollutants will be discharged to the WWTP effluent. The only burdens in this dataset stem from infrastructure, pumping energy, and the disposal of grit waste (see chapter 4.7 'Process-specific burdens' on page 32). This dataset can be compared to the treatment of average sewage in a plant of equal size (second dataset in Tab. 7.1). The difference between these two modules is caused by the presence of pollutants in the sewage, which lead to additional expenditures for auxiliary treatment materials, from treatment energy and disposal of for generated digester sludge. A large number of burdens are significantly depending on the 'baseline' burden established by 'unpolluted sewage'. Only BOD to water and cadmium to soil are largely depending on sewage pollutants contents (BOD in effluent from WWTP, and cadmium to sludge landfarming).

As explained above BOD emissions to water are mainly depending on sewage composition. This is the reason the BOD emissions in the first three datasets is almost identical. Only a small share of BOD emissions originate from background processes. The cadmium emissions *to soil* are also depending on sewage composition, but also on applied sludge disposal type. In small scale WWTPs more sludge is landfarmed (see Tab. 5.2 on page 37). This significantly increases the cadmium emissions to soil for treatment in small scale WWTPs. The applied sludge disposal type is based on Swiss averages but can be changed by the user in the WWTP calculation tool.

7.3 Relevance of volatile metalocenes in digester gas

In this study volatilisation of Cd, Pb, As, Hg, Sb, and Sn during sludge digestion was inventoried, cf. section 'Volatile metals' on page 29. During the inventoried process chain WWTP→digester→sludge→incinerator there is only one other process where metal air emissions occur: during the disposal of sludge in an incinerator. For average wastewater in a class 3 WWTP with 53.1 w-% sludge incineration the emissions of tin, lead and cadmium are dominated by emissions from sludge incineration. On the other hand, the emissions of *mercury, arsenic and antimony* are strongly dominated by the emissions from digester gas. For wastewaters containing these pollutants the volatilisation could be a relevant damage pathway. Apart from average wastewater (with As and Hg) in the current inventory also concrete production effluent (with As) and glass production effluent (with As and Sb) contain such pollutants (cf. part I on waste compositions). In the cumulated inventory data the relevance of these emissions from digester gas are reduced, as sources of mercury, arsenic and antimony air emissions from background process are added.

8 Conclusions

The waste-specific inventory of wastewater treatment shows significant discrimination between different sewage compositions. Inventorying wastewater treatment without heeding the specific pollutant contents leads to oversimplification. Very significant or even dominant burdens also originate from the infrastructure. Wastewater treatment plants earn a rather unique place in LCA in that respect, as in most other services and processes infrastructure is not as relevant as here. Neglecting infrastructure in wastewater treatment will lead to significant assessment gaps.

Apart from average sewage, only few pollutant data is known for sewage of production processes. Assessment gaps are likely to occur here as well. In the future better wastewater composition data could be targeted.

The influence of the size or capacity class of the wastewater treatment plants is also significant. However often the capacity class applicable to a certain wastewater had to be estimated.

The vector of considered elements has been extended especially for metals. The transfer coefficients of metals are rather crude and better data, based on measurements, would be desirable. Uptake of dissolved metals into sludge biomass should be investigated regarding its relevance.

The present model is only applicable for municipal wastewater treatment, but not for industrial wastewater treatment, that treats wastewaters rich in e.g. metals or cyanide. A model for industrial wastewater treatment could be created, if necessary. A starting point could be the treatment of flue gas scrubber liquid in waste incinerators (cf. conclusions in part II).

Abbreviations

a	annum = year, used in expressions like "kg/a" for "kilogram per year".
ARA	Swiss German expression for (municipal) wastewater treatment plant ('Abwasser-Reinigungs-Anlage'). (German: 'Kläranlage').
BOD	Biological oxygen demand (German 'Biologischer Sauerstoffbedarf, BSB'). A measure for the content of readily degradable hydrocarbons. Equates to the amount of oxygen taken up in microbial decomposition of hydrocarbons – usually in dark incubation at 20°C and over 5 days (BOD_5). Given as mass of absorbed oxygen (O_2). Usually smaller than COD.
COD	Chemical oxygen demand (German 'Chemischer Sauerstoffbedarf, CSB'). A measure for the total content of degradable hydrocarbons. Equates to the amount of oxygen taken up in oxidation of hydrocarbons using a oxidation agent like potassium permanganate $KMnO_4$ or potassium dichromate $K_2Cr_2O_7$. Given as mass of absorbed oxygen (O_2). Usually larger than BOD.
DOC	Dissolved organic carbon. Given as mass of carbon (C_{org}). Smaller or equal than TOC (exludes organic carbon in undissolved solids and particulates).
EAWAG	Swiss Federal Institute for Environmental Science and Technology (German 'Eidgenössische Anstalt für Wasserversorgung, Abwasserreinigung und Gewässerschutz').
FAL	Swiss federal research station for agroecology and agriculture, Reckenholz, Zürich (Eidgenössische Forschungsanstalt für Agrarökologie und Landbau)
GSD	(also SDg) Geometric Standard Deviation. Uncertainty measure for lognormally distributed values. The GSD cannot be smaller than 1 (i.e. 100%). The squared GSD value (GSD^2) can be used to calculate 95% confidence intervals around the mean. More information on the uncertainty calculations in ecoinvent 2000 can be found in part I and the methodology report (Frischknecht et al. 2007).
IPCC	Intergovernmental Panel on Climate Change

l	lower (2.5%) confidence value
m	mean value
MSWI	Municipal Solid Waste Incinerator (Swiss expression: KVA Kehrichtverbrennungsanlage, German expression: MVA Müllverbrennungsanlage)
PCE	Per-captia-equivalents. Unit used to express WWTP capacities. Also called 'inhabitant units', German 'Einwohnergleichwerte' EW or EGW. One PCE equates to 60 g BOD/day or 22 kg BOD/year, which is the typical BOD load generated by one person (BUWAL 1996:9).
SKN	SKN signifies the soluble Kjeldahl nitrogen (German: gelöster Kieldahl-Stickstoff GKN). Kjeldahl nitrogen is a sum parameter for organic bound nitrogen (N_{org}) and ammonium (NH_4^+).
TKN	TKN signifies the total (soluble + particulate) Kjeldahl nitrogen. Kjeldahl nitrogen is a sum parameter for organic bound nitrogen (N_{org}) and ammonium (NH_4^+).
TOC	Total organic carbon. Given as mass of carbon (C_{org}). Larger or equal than DOC, since it includes also carbon in undissolved solids or particulates.
u	upper (97.5%) confidence value
WWT	(municipal) wastewater treatment
WWTP	(municipal) wastewater treatment plant. Also called effluent treatment plant or sewage treatment plant (German: 'Kläranlage', Swiss German: 'Abwasser-Reinigungs-Anlage').
ZH	Canton of Zürich, Switzerland.

Glossary of terms

Ammonium	Ammonium is the cation NH_4^+. At high pH it can be converted to neutral ammonia NH_3, which is poisonous to water organisms. At a pH of 7 most ammonia is present as ammonium NH_4^+.
Class	Capacity class or size class (German 'Grössenklassen'). Coarse grouping of Swiss WWTPs and associated sewer systems according to the treatment capacity. See Tab. 2.1 on page 2. Capacity classes are based on the treatment capacity expressed in per-captia-equivalents PCE.
Effluent	The *output* of the WWTP, discharged to a natural water body. Identical to 'discharge'. The *input* into a WWTP (which might in turn be the effluent of a household or a production site) is called 'wastewater' or 'sewage'.
Kjeldahl	sum parameter for organic bound nitrogen (N_{org}) and ammonium (NH_4^+). SKN signifies the soluble Kjeldahl nitrogen. TKN signifies the total (soluble + particulate) Kjeldahl nitrogen.
Sewage	Input to the WWTP, transported in sewers. Identical to 'wastewater'. The output of the WWTP is called 'effluent' or 'discharge'.

9 Appendix

Tab. A. 1 Raw data for WWTP infrastructure from (Flückiger & Gubler 1994, Fahrner et al. 1995) cited in (Zimmermann et al. 1996:C.21)

Infrastructure		WWTP Twann	WWTP Werdhölzli	WWTP Ergolz
Inhabitant Equivalents	PCE	1'600	593'300	25'000
Concrete	kg/PCE*a	67.54	11.94	65.34
Reinforcement steel	kg/PCE*a	2.11	0.34	2.04
Steel	kg/PCE*a	0.08	0.014	0.077
Drinking water	kg/PCE*a	4.16	0.72	4.02
Electricity	kWh/PCE*a	2.69E-05	4.72E-06	2.61E-05
Excavation	m^3/PCE*a	0.067	0.012	0.064

10 References

AGW 1993 — AGW (1993) Deponienachsorge Schlussbericht. Sieber Cassina + Partner AG und Gerling Consulting Gruppe (Autoren) im Auftrag das Amtes für Gewässerschutz und Wasserbau des Kantons Zürich.

ARB 2001 — ARA Region Bern (2002) Technischer Bericht 2001. ARA Region Bern, Herrenschwanden BE, Switzerland. Retrieved January 16, 2003 from http://www.ara-bern.ch/d/pdf/tech_bericht_01.pdf

Belevi & Baccini 1989 — Belevi H., Baccini P. (1989): Long-term Behaviour of Municipal Solid Waste Landfills. Waste Management & Research, Vol. 7, p. 43-56.

Boller & Häfliger 1996 — Boller M., Häfliger M. (1996) Verbleib von Schwermetallen bei unterschiedlicher Meteorwasserentsorgung. Gas Wasser Abwasser, No.1/1996. Zürich, Switzerland.

BUWAL 1994 — BUWAL (1994) Daten zum Gewässerschutz in der Schweiz. Umwelt-Materialien Nr. 22, Gewässerschutz, BUWAL, Berne, Switzerland. Retrieved January 17, 2002 from http://aumwww.unibe.ch/buwal/dokugs4/um22d.pdf

BUWAL 1996 — BUWAL (1996) Stickstofffrachten aus Abwasserreinigungsanlagen. Schriftenreihe Umwelt Nr. 276, BUWAL, Berne, Switzerland. Retrieved January 17, 2002 from http://aumwww.unibe.ch/buwal/dokugs6/srud276.pdf

BUWAL 1997 — BUWAL (1997) Umwelt in der Schweiz – Kapitel 11 Wasser. Bundesamt für Umwelt, Wald und Landschaft and Bundesamt für Statistik BfS, Berne, Switzerland. Retrieved April 8, 2002 from http://www.buwal.ch/d/themen/umwelt/wasser/dk11u00.pdf

BUWAL 2001a — BUWAL (2001) Entsorgung des Klärschlammms 2000, Stand 11.12.2001, BUWAL, Berne, Switzerland. Retrieved March 2, 2002 from http://www.buwal.ch/abfall/docu/pdf/tab15_d.pdf

BUWAL 2001b — BUWAL (2001) Datenerhebung für den Vollzug des Gewässerschutzgesetzes Stand 01.01.2000. BUWAL, Berne, Switzerland. Retrieved January 17, 2002 from http://aumwww.unibe.ch/buwal/dokugs1/AbwStat01_d_AbwasserentsorgungStatistik2001.pdf and http://aumwww.unibe.ch/buwal/dokugs1/AbwStat01_d_AbwasserentsorgungStatistik2001.xls

BUWAL 2002a — BUWAL (2002) Aus für den Klärschlamm in der Düngung. Press release of BUWAL May 13, 2002 regarding the changes in the hazardous substance ordinance (Verordnung über umweltgefährdende Stoffe). BUWAL, Berne, Switzerland. Retrieved July 27, 2002 from http://www.umwelt-schweiz.ch/buwal/de/medien/presse/artikel/20020513/00600/index.html

BUWAL 2002b — BUWAL (2002) Abwasserreinigung in der Schweiz am 1. Januar 2001. BUWAL, Berne, Switzerland. Retrieved July 27, 2002 from http://www.umwelt-schweiz.ch/imperia/md/content/gefisch/abwland/abwasserRch2001.pdf.

BUWAL 2002c — BUWAL (2002) Bestehende Datenbanken, die zur Zeit für die Integration in GIS/GEWISS vorbereitet werden. Abteilung Gewässerschutz + Fischerei, BUWAL, Berne, Switzerland. Retrieved http://www.umwelt-schweiz.ch/imperia/md/content/gefisch/gis/GISDBsG+F.pdf, See also http://www.umwelt-schweiz.ch/buwal/de/info/buwal/organisation/abteilungen/abt_gewaesser/gfinforubrik_4/GISGF/unterseite9/index.html

Crinum 2002 — Crinum AG (2002) Mikroorganismen der Abwasserreinigung. Pictures taken December 28, 2002 from http://www.crinum.de/abwasser/mikroorganismen.html

DBGS 1994 — Internal excerpts of the water protection database (Datenbank Gewässerschutz) of BUWAL (1994) Referring to data of 1994. Cited in Zimmermann et al. 1996:C.9ff.

EEA 2002 — Brodersen J., Juul J., Jacobsen H., Tsotsos D. (2002) Review of selected waste streams: Sewage sludge, construction and demolition waste, waste oils, waste from coal-fired power plants and biodegradable municipal waste. European Topic Centre on Waste, European Environment Agency, Copenhagen, Denmark. Retrieved March 11, 2002 from http://reports.eea.eu.int/technical_report_2001_69/en/tech_rep_69.pdf

Engitech 2002 — Engitech (2002) Activated Sludge Microorganisms. Engitech Inc., Texas, USA. Pictures taken December 28, 2002 from http://www.engitech.com/asm.htm

Fahner et al. 1995 — Fahner S., Bührer H., Grabski C. (1995) Ökobilanz einer kommunalen ARA am Beispiel der ARA 'Ergolz I' in Sissach, BL. Diploma work Ingenieurschule beider Basel, Muttenz.

Feldmann & Hirner 1995 — Feldmann J., Hirner A.V. (1995) Occurrence of volatile metal and metalloid species in landfill and sewage gases. Intern.J.Environ.Anal.Chem. 60 (1995) 339-359

Flückiger & Gubler 1994 — Flückiger P. & Gubler D. (1994) Grundlagen für die Integration des Downstreamprozesses Abwasserreinigung in Ökobilanzen. Analyse von zwei Fallbeispielen ARA Twann/Ligerz/Lüscherz und ARA Werdhölzli/Zürich. Diploma work ETH Zürich.

Frischknecht 2000 — Frischknecht R. (2000) Allocation in Life Cycle Inventory Analysis for Joint Production. Int. J. LCA, Vol. 5, Nr. 2. Online-First: February 2000. Retrieved July 28, 2002 from www.sylvatica.com/LCA_2_2000_Frischknecht.pdf or http://dx.doi.org/10.1065/lca2000.02.013

Frischknecht et al. 1996 — Frischknecht R., Bollens U., Bosshart S., Ciot M., Ciseri L., Doka G., Dones R., Gantner U., Hischier R. and Martin A. (1996): Ökoinventare von Energiesystemen. ETH/PSI, Zürich/Villigen.

Frischknecht et al. 2007 — Frischknecht R., Jungbluth N., Althaus H.-J., Doka G., Dones R., Hischier R., Hellweg S., Nemecek T., Rebitzer G. and Spielmann M. (2007) Overview and Methodology. Final report ecoinvent data v2.0, No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf, CH, Online-Version under: www.ecoinvent.ch.

Fritsche 1985 — Fritsche W. (1985) Umwelt-Mikrobiologie. Akademie-Verlag Berlin, Germany.

GALU 2001 — Gemeindeverband Abwasserreinigung der Region Luzern (2001) Images of WWTP Buholz, Emmen LU, retrieved November 28, 2002 from http://www.ara-luzern.ch

GSchV 2001 — Swiss national law (2001) Gewässerschutzverordnung (GSchV) vom 28. Oktober 1998 (Stand am 18. Dezember 2001). Retrieved January 7, 2003 from http://www.admin.ch/ch/d/sr/8/814.201.de.pdf

Guinée et al. 2001 — Guinée J.B. (Ed.), Gorrée M., Heijungs R., Huppes G., Kleijn R., de Koning A., van Oers L., Wegener Sleeswijk A., Suh S., Udo de Haes H.A., de Bruijn H., van Duin R., Huijbregts M.A.J., Lindeijer E., Roorda A.A.H., van der Ven B.L., Weidema B.P. (2001) Life Cycle Assessment – a operational guide to the ISO standards – part 2a Guide. Minisry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Leiden

University, Netherlands. Retrieved May 17, 2001 http://www.leidenuniv.nl/interfac/cml/lca2

Gujer 1993 — Gujer. W. (1993) Grundzüge der Siedlungswasserwirtschaft. Vorlesungsskript WS 1993/94, Institut für Hydromechanik und Wasserwirtschaft, ETH Zürich.

Gujer 1995 — Gujer. W. (1993) Biologische Abwasserreinigung. Vorlesungsskript SS 1995, Institut für Hydromechanik und Wasserwirtschaft, ETH Zürich.

Herter et al. 2001 — Herter U., Külling D., Becker van Slooten K., Brüschweiler B., Doherr M., Frossard R., Heim D., Hett A., Huguenin O., Kupper T., Perler L., Schmidt J., Widmer F., Zarn J. (2001) Risikoanalyse zur Abfalldüngerverwertung in der Landwirtschaft – Teil 1: Grobbeurteilung. Eidgenössische Forschungsanstalt für Agrarökologie und Landbau FAL, Reckenholz, Zürich, Switzerland. Retreived February 27, 2002 from http://www.blw.admin.ch/fakten/texte/d/risikoanavoll.pdf

Hobson et al. 2000 — Hobson J., Johnke B. et al. (2000) IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories – Chapter 5 Waste. Intergovernmental Panel on Climate Change IPCC. Retreived March 4, 2002 from http://www.ipcc-nggip.iges.or.jp/public/gp/pdf/5_Waste.pdf

Huijbregts 1999 — Huijbregts M. (1999) Life-cycle impact assessment of acidifying and eutrophying air pollutants – Calculation of equivalency factors with RAINS-LCA. Interfaculty Department of Environmental Science, Faculty of Environmental Sciences, University of Amsterdam, Netherlands. Retrieved March 22, 2000 from http://www.leidenuniv.nl/interfac/cml/lca2/Report_MH_IIASA2.pdf

Hyperion 2002 — Hyperion (2002) Micro-organisms and their role in the Activated-Sludge Process. Hyperion Treatment Plant, Playa Del Rey, California, USA. Pictures taken December 28, 2002 from http://www.college.ucla.edu/webproject/micro7/studentprojects7/Rader/asludge2.htm

Jungbluth 2001 — Jungbluth N. (2001) Berechnungstool der Eingabewerte für Wasserparameter und Einzelstoffe – Anhang zu den Qualitätsrichtlinien, Excel file 'NJ csb-tool-1.0 2 .xls' of December 16, 2001

Kaufmann 2001 — Kaufmann U. (2001) Schweizerische Statistik der erneuerbaren Energien 2000. Dr.Eicher+Pauli AG, Liestal, Switzerland. Comissioned by Bundesamt für Energie BfE, Berne, Switzerland. Retrieved April 8, 2002 from http://www.energie-schweiz.ch/imperia/md/content/teilstatistiken/6.pdf

Kellenberger et al. 2007 — Kellenberger D., Althaus H.-J., Jungbluth N. and Künniger T. (2007) Life Cycle Inventories of Building Products. Final report ecoinvent data v2.0 No. 7. Swiss Centre for Life Cycle Inventories, Empa - TSL, Dübendorf, CH, Online-Version under: www.ecoinvent.ch.

KfU 1992 — Koordinationsstelle für Umweltschutz (1992) Umweltbericht für den Kanton Zürich 1992. Direktion der öffentlichen Bauten (Ed.) im Auftrag des Regierungsrates des Kantons Zürich.

Koppe & Stotzek 1993 — Koppe P., Stotzek A (1993) Kommunales Abwasser. Vulkan Verlag, Essen, Germany.

Külling et al. 2002 — Külling D. R., Stadelmann F. X., Candinas T., (2002) Nährstoffe und Schwermetalle im Klärschlamm 1975 - 1999. Agrarforschung, 9 (5): 200-205.

Labhardt 1996 — Labhardt A. (1996) Ausführliche Inventarisierung des Systems Kanalisation. ESU-Reihe, 2/96, Gruppe Energie-Stoffe-Umwelt ESU, Laboratorium für Energiesysteme, Institut für Energietechnik, ETH Zürich, Switzerland.

Liebi 2002 — Liebi C. (2002) Umweltbericht 2001 Kläranlage Kloten/Opfikon. Glattbrugg, Switzerland. Retrieved December 28, 2002 from http://www.klaeranlage.ch/download/Umweltbericht%202001.pdf

MacDonald 1994 — MacDonald D.D. (1994), Approach to the Assessment of Sediment Quality in Florida Coastal Waters; Volume 1 - Development and Evaluation of Sediment Quality Assessment Guidelines. Prepared by MacDonald Environmental Sciences Ltd., Ladysmith, British Columbia for Florida Department of Environmental Protection Office of Water Policy, Florida, USA. Retreived March 4, 2003 from http://www.dep.state.fl.us/water/monitoring/docs/seds/vol1/volume1.pdf; see also http://www.dep.state.fl.us/water/monitoring/seds.htm

Menzi et al. 1997 — Menzi H., Frick R., Kaufmann R. (1997) Ammoniak-Emissionen in der Schweiz: Ausmass und technische Beurteilung des Reduktionspotentials. Schriftenreihe der FAL 26, Zürich-Reckenholz.

Michel 1938 — Michel R. (1938) Berechnung der Verbrennungswärmen fester und flüssiger Brennstoffe nach den Wärmewerten ihrer Einzelbestandteile. Feuerungstechnik 26, No.9, p.273-278 (Leipzig, Germany. Appeared 1912 –1943).

Müller et al. 1994 — Müller E., Thommen R., Stähli P. (1994) Energie in ARA. Bundesamt für Energiewirtschaft, Bundesamt für Konjunkturfragen und Bundesamt für Umwelt, Wald und Landschaft (Hrsg.), Berne, Switzerland.

Nemecek et al. 2007 — Nemecek T., Kägi T. and Blaser S. (2007) Life Cycle Inventories of Agricultural Production Systems. Final report ecoinvent v2.0 No.15. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, Online-Version under: www.ecoinvent.ch.

OECD 2001 — OECD (2001) Key Environmental Indicators, OECD Environment Directorate. Retrieved February 17, 2002 from http://www.oecd.org/pdf/M00019000/M00019613.pdf

Schmid et al. 2000 — Schmid M., Neftel A., Fuhrer J. (2000) Lachgasemissionen aus der Schweizer Landwirtschaft. FAL, Schriftenreihe der FAL 33, 131 p.

Spielmann et al. 2003 — Spielmann M., Kägi T. and Tietje O. (2003) Life Cycle Inventories of Transport Services. Final report ecoinvent 2000 No. 14. Swiss Centre for Life Cycle Inventories, ETH-Zurich, UNS, Dübendorf, CH, Online-Version under: www.ecoinvent.ch.

Stadelmann et al. 2002 — Stadelmann F.X., Külling D., Herter U. (2002) Klärschlamm – Dünger oder Abfall. EAWAG news Nr. 53, EAWAG Dübendorf, Switzerland. Retrieved August 21, 2002 from http://www.eawag.ch/publications/eawagnews/www_en53/en53d_printer/en53d_stadelm_p.pdf

StoV 1999 — Swiss national law (1999) Verordnung vom 9. Juni 1986 über umweltgefährdende Stoffe, Stoffverordnung StoV (Stand am 1.1.1999). SR 814.013. EDMZ, Berne, Switzerland.

Thum 1995 — Thum D. (1995) Daten zur ARA Ottenbach-Jonen, Ottenbach ZH, Switzerland

UAG 1999 — Umwelt Aargau (1999) Erfolgreiches Betriebsjahr 1998 für die kommunalen ARA. Umwelt Aargau Nr. 7 (Oktober) p.15-18, Retrieved January 17, 2002 from http://www.ag.ch/umwelt-aargau/pdf/UAG_7_15.pdf

von Raczeck 1993 — von Raczeck, H. (1993) Experimentelle Untersuchung des Emissions- und Abbrandverhaltens von Klärschlämmen in einer halbtechnischen Wirbelschichtfeuerung, VDI-Berichte, Reihe 6: Energieerzeugung, Nr. 281, Düsseldorf.

Zimmermann et al. 1996 — Zimmermann P., Doka G., Huber F., Labhardt A., Menard M. (1996): Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen. ESU-Reihe, 1/96, Gruppe Energie-Stoffe-Umwelt ESU, Laboratorium für Energiesysteme, Institut für Energietechnik, ETH Zürich, Switzerland. This source also contains data from the unpublished BUWAL database in Swiss WWTP of 1994.