Final report for subtask 5 of the demonstration project
"Sanitation Concepts for Separate Treatment
of Urine, Faeces and Greywater " (SCST)

financed by: *Kompetenzzentrum Wasser Berlin gGmbH*
Anjou Recherche, Paris
co-financed by: *the European Commission*
(LIFE 03ENV/D/000025)

Ecological assessment of alternative sanitation concepts with Life Cycle Assessment

Technische Universität Berlin
Institute of Environmental Technology
Department of Water Quality Control

Project coordinator: *Prof. Dr.-Ing. Martin Jekel*
Authors: *Dipl.-Ing. Christian Remy*
Dr.-Ing. Alexander Ruhland

Berlin, December 2006

Abstract

The goal of this study is the identification of ecological advantages and disadvantages of alternative sanitation systems in comparison to conventional wastewater treatment. The methodology of Life Cycle Assessment (LCA) is adopted as an evaluation tool for the ecological assessment of various sanitation scenarios for a hypothetical middle-sized settlement in Germany (ca 5000 inhabitants). The scenarios include a reference system with conventional drainage and treatment in an activated sludge plant with anaerobic sludge digestion and sewage gas production. In the alternative scenarios, urine is source-separated in the toilet, collected and applied as fertilizer. Faeces are either collected by gravity drainage and composted together with biowaste or collected by a vacuum system and co-digested with biowaste to gain biogas for energy production. The remaining greywater is treated in a soil filter or in a technical plant (Sequencing batch reactor).

All relevant processes of the investigated scenarios are modelled in detail for the Life Cycle Inventory, based on data from pilot plants and literature. This implies the processing of the different waste fractions, transport and energy supply, mineral fertilizer substitution, and sludge incineration. Beside the operational expenditures, the construction phase is included with material and energy demands. The resulting substance flow model is evaluated with a set of environmental indicators relating to the demand of energy, non-renewable resources, climate change, eutrophication, acidification, and various toxicity potentials.

As a result, the alternative scenarios cause less environmental burden in almost all impact categories. The source-separation of human excreta disburdens the wastewater treatment process and lowers nutrient emissions into surface waters. The secondary fertilizer from urine and faeces has lower heavy metal content than an average mineral fertilizer. Depending on the system configuration, alternative sanitation systems can have a lower demand for fossil fuels and subsequently cause fewer emissions of climate-active gases. Only the increased emission of acidifying gases represents a considerable drawback compared to the conventional system. A normalisation of all indicators to the average environmental burden of a single person in Germany reveals that the decisive categories for the overall comparison are related to eutrophication, acidification, and terrestrial ecotoxicity. Energy-related indicators have a smaller contribution, but they can be important in terms of world-wide scarce fossil resources and climate change. The advantages of alternative sanitation systems can only be realized if the secondary functions of mineral fertilizer substitution and energy supply are fully utilized.

Important key parameters for future LCA studies of alternative sanitation systems are identified, which may simplify the data acquisition. The construction phase has only a minor relevance for the ecological assessment and may therefore be neglected in future studies. In all, the data quality of this LCA study can be further improved, because many processes of alternative systems have not yet been investigated or realized in full-scale. Hence, the development of a universal decision support method could not be realized in a reasonable way due to the lack of adequate long-term process data and the high influence of case-specific boundary conditions on the technical implementation. However, this LCA study gives a first assessment of potential ecological benefits and drawbacks of alternative sanitation systems.

Contents

1 Introduction

In the scope of the EU demonstration project “Sanitation concepts for separate treatment of urine, faeces and greywater” (SCST), more experience should be gained with the implementation and operation of alternative sanitation concepts. These new sanitary systems are based on the separate collection and treatment of the different wastewater fractions and the recycling of nutrients from human excreta to agriculture. Within the SCST project, potential processes of alternative sanitation systems are investigated in pilot-scale to generate reliable and detailed information concerning design, planning, operation and maintenance of the new sanitation systems. Beside the technical realisation, an assessment of the ecological and economical aspects of the new systems should quantify the related benefits or drawbacks.

The present report describes the ecological assessment of the investigated alternative sanitation concepts with the methodology of Life Cycle Assessment (LCA). The integrated approach of this evaluation method allows a comprehensive analysis and evaluation of the ecological aspects of technical systems. More precisely, the goals of this LCA are to:

- identify ecological advantages of new sanitation concepts in relation to the conventional sanitation system
- reveal ecological hotspots of new sanitation concepts
- determine relevant key parameters and processes for an LCA assessment of sanitation concepts, thus facilitating future LCA studies in this field.

An important starting point for the systematic analysis of the sanitation systems is a precise definition of the goal and scope of the LCA. In chapter 2, the boundary conditions of this LCA study are defined together with a detailed description of the analysed systems. The evaluation method for Life Cycle Impact Assessment (LCIA) is presented with its impact categories and related indicators.

Chapter 3 describes the Life Cycle Inventory (LCI), i.e. the process model for all relevant parts of the conventional and alternative sanitation systems. A careful documentation of the data sources and assumptions made in these models is essential for the plausibility of the LCA results.

Chapter 4 presents the results from LCIA for the environmental indicators and for overall eco-profiles of the compared systems. The stability of the results in relation to important system parameters is further investigated in sensitivity analysis.

Finally, chapter 5 summarizes the results of this LCA study and draws the conclusions. The ecological benefits and drawbacks of the different processes in the investigated alternative sanitation systems are summarized. Important key parameters for an LCA assessment of sanitation systems are listed to allow simplified LCA studies in the future.

2 Basics of Life Cycle Assessment and methodological definitions

For comparative ecological studies meanwhile the tool "Life Cycle Assessment" (LCA) became accepted and is widely used as it is roughly defined in ISO 14040, 1997). LCA aims for evaluating the environmental burdens associated with a product, process or service by quantifying all material and energy flows linked with the analysed economic activity. The assessment includes the entire economic system that is necessary to fulfil the economic activity, encompassing extracting and processing raw materials, production, transportation, use, recycling and final disposal. A detailed description and discussion of LCA-models include for instance Guinée et al., 2002 or Frischknecht, 1998.

Figure 1 shows the simplified structure of an LCA system with the production process or service under investigation, the preceding and subsequent processes, and input and output streams crossing the system boundary.

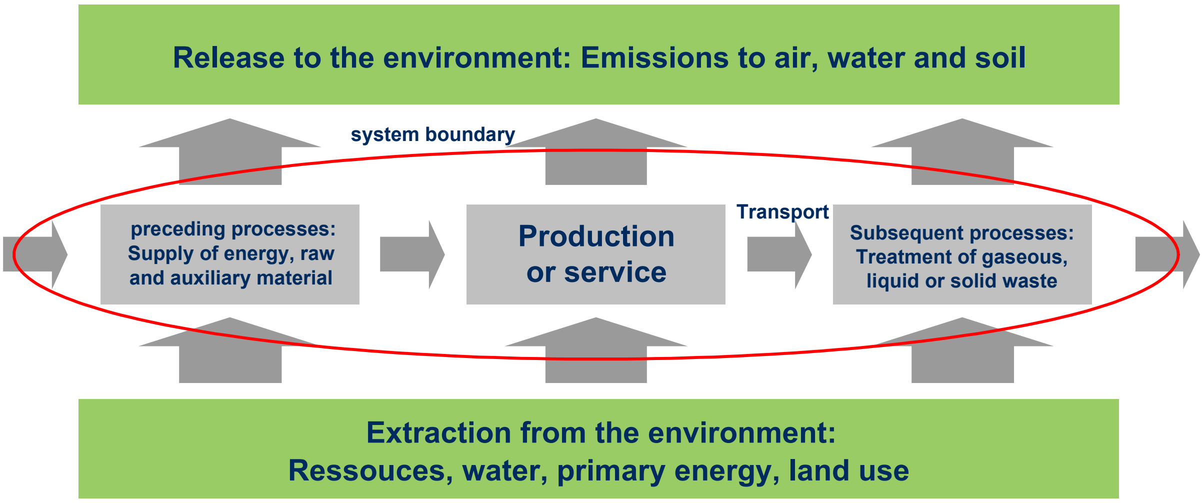


Figure 1: Simplified structure of an LCA framework

ISO 14040, 1997 defines the following elements of an LCA study that are treated iteratively:

- Goal and scope definition
- Inventory analysis
- Impact assessment and
- Interpretation

This study follows the structure and the main requirements of ISO 14040 ff. However, it does not cover all requirements like a peer review that is obligatory for comparative studies according to ISO. The following chapters comprise the most important methodological definitions for goal and scope definition, inventory analysis, life cycle impact assessment and interpretation of the results, and finally necessary assumptions and limitations of the LCA case study.

2.1 Goal and scope definition

2.1.1 Goal and target group

This study analyses a typical conventional sanitation concept (CS system) and four new sanitation concepts for separate treatment of urine, faeces, greywater and biowaste (SCST systems) in order to compare the systems with regard to their ecological impacts on the environment. The goal is to identify advantages, disadvantages and ecological hotspots of the different sanitation concepts, so that these could be taken into account during further development and planning of SCST systems. Depending on the assumed boundary conditions, the ecologically preferable sanitation concept shall be revealed. Key parameters with high relevance for the ecological system comparison will be established, so that future LCA studies of sanitation systems may use a simplified approach and less extensive data acquisition.

No existing marketable products are compared, but integrated concepts of urban drainage and disposal of biowaste. The results of the study are intended for experts from research and innovative companies in order to expand the knowledge in the field of integrated municipal disposal concepts and to get hints to improve existing approaches and future developments.

2.1.2 Function and functional unit

The primary function of the analysed systems is to fulfil the services:

- *removal and disposal of human urine and faeces from households,*
- *drainage and treatment of waste water, charged with substances resulting from domestic washing machines, kitchen residuals and water from personal hygiene (greywater) and*
- *disposal of solid biowaste resulting from kitchen, garden and municipal greens.*

The material and energy flows associated with the above mentioned services, respectively the LCA results, are related to the functional unit, which is defined as

the performance of the above mentioned services for one person during one year.

Beside the primary functions of the investigated systems, secondary functions like the supply of fertilizing substances (e.g. from urine or faeces) or energy (e.g. from digestion and biogas use) in SCST systems are coupled with the primary services. For proper system comparison, those secondary functions are taken into account by the method of system expansion (see 2.1.4).

As a further boundary condition for the study, it is assumed that the respective sanitation systems shall be installed in a middle-sized settlement in Germany (ca. 5,000 inhabitants). Due to available GIS data from a certain settlement area for the construction inventory, the respective number of inhabitants is set to 4891 individuals. This equals exactly the current population of this area in Berlin-Nikolassee.

2.1.3 Reference input flows

For the reference input flows of the different wastewater and waste streams, a typical pattern of Western Europe countries is considered (Table 1). Whereas the amount and quality of human urine and faeces are relatively well-known, greywater volume and composition may vary over wide ranges, depending on regional conditions and user behaviour. Representing the decreasing water consumption in Germany, this study assumes an average amount of 80 L of greywater per person and day.

From the biowaste streams, especially loppings show a wide variety during the seasons of a year with regard to quantity and composition. The structure of the urban area, the share of garden area and municipal greens define the quantity and quality of loppings. The value 0.3 kg/(P*d) wet mass represents the average potential of organic garden waste in Germany. In addition there is an average potential of 0.2 kg/(P*d) wet mass for municipal greens (Wintzer et al., 1996). The range of the mass flow of mixed urban biowaste during one year is about factor 2, the range of the volume flow about factor 3 (Fricke, 1990).

Decisive for the environmental characteristics is the composition of the waste flows shown in Table 2 and Table 3. Drinking water is used as flushing water in all investigated systems (no water reuse is considered), so flush water composition is listed in as well. It must be emphasized that all assumed quantities and the composition represent average values and are subject to change. A comprehensive data collection is provided in the annex.

Table 1: Reference input mass flows (wastewater and waste fractions)

Flows	Per person and day	Per person and year	Settlement (4891 inh.)	Remarks
	[kg/P*d]	[kg/P*a]	[t/a]	
Urine	1.5	547.5	2678	
Faeces	0.14	51.1	250	wet mass
Greywater	80	29,200	142,817	
Organic kitchen residuals	0.16	58.4	286	wet mass
Loppings/ garden waste	0.3	109.5	536	wet mass

Table 2: Average composition of faeces, urine, greywater and biowaste from kitchen

	unit	urine	faeces	greywater*	org. kitchen residuals
Quantity	kg/(P d)	1.50	0.14	80.00	0.16
Main constituents and nutrients					
dry matter	mg/(P d)	60,000	45,000	120,000	50,000
organic dry matter	mg/(P d)	45,000	42,000		36,000
COD	mg/(P d)	15,000	35,000	60,000	
TOC	mg/(P d)	7,000	21,000	18,000	13,000
N-total	mg/(P d)	10,000	1,500	1,300	900
P-total	mg/(P d)	1,000	500	500	200
K	mg/(P d)	2,600	550	2,000	600
Na	mg/(P d)	3,500	150	6,000	1.200
Ca	mg/(P d)	210	1,000	14,000	1.000
Mg	mg/(P d)	120	200	3,000	220
Cl	mg/(P d)	4,800	60	7,000	3,000
S-total	mg/(P d)	800	200	7,500	100
Metals					
Cd	mg/(P d)	0.0002	0.02	0.20	0.01
Cr	mg/(P d)	0.01	0.02	3.00	0.50
Cu	mg/(P d)	0.05	1.50	20.00	1.00
Hg	mg/(P d)	0.0004	0.02	0.02	0.01
Ni	mg/(P d)	0.04	0.20	2.00	0.20
Pb	mg/(P d)	0.01	0.02	3.00	0.60
Zn	mg/(P d)	0.25	10.00	46.00	7.30

* from kitchen, washing machine, and personal hygiene; including loads from tap water with a typical ionic composition as well as copper and zinc loads after contact with drinking water pipes

Table 3: Average composition of loppings and flush water

	unit	loppings	unit	flush water
Quantity	kg d.m./(P d)	0,30	kg/(P d)	depends on system
Main constituents and nutrients				
dry matter	% wet mass	41	mg/L	520
organic dry matter	g/kg dry matter	710	mg/L	
TOC	g/kg dry matter	370	mg/L	1.10
N-total	g/kg dry matter	11	mg/L	1.00
P-total	g/kg dry matter	5	mg/L	0.08
K	g/kg dry matter	13.6	mg/L	7.50
Na	g/kg dry matter	0.2	mg/L	36.00
Ca	g/kg dry matter	33	mg/L	103.00
Mg	g/kg dry matter	4.5	mg/L	10.00
Cl	g/kg dry matter	0.3	mg/L	18.00
S-total	g/kg dry matter	0.5	mg/L	40.50
Metals				
Cd	mg/kg dry matter	0.40	mg/L	0.0005
Cr	mg/kg dry matter	4.60	mg/L	0.005
Cu	mg/kg dry matter	19.00	mg/L	0.16
Hg	mg/kg dry matter	0.20	mg/L	0.0002
Ni	mg/kg dry matter	3.70	mg/L	0.005
Pb	mg/kg dry matter	4.80	mg/L	0.005
Zn	mg/kg dry matter	110.00	mg/L	0.37

2.1.4 System expansion

Depending on the design, SCST systems may produce additional products beside the disposal service, like energy generated by use of biogas in combined heat and power units or fertilizers produced by nutrient recycling. Usually, different SCST and CS systems generate different amounts of additional products or services. Therefore the systems fulfil different functions and thus are not comparable on an equivalent base of reference. In order to establish the same base of reference, the systems have to be expanded by the corresponding deficient amount of equivalent products. This procedure is called system expansion and is described in detail for instance in ISO/TR 14049 (2000) and Fleischer (1994).

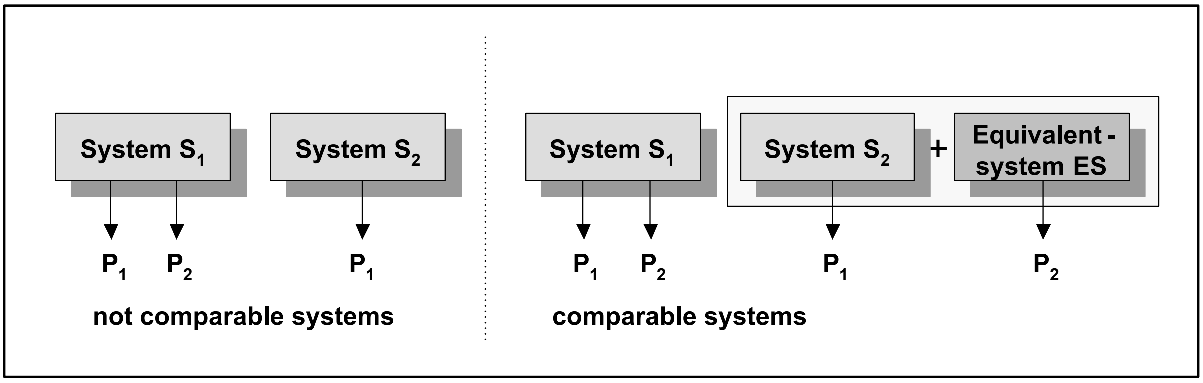


Figure 2: Principle of system expansion

The basic procedure of system expansion is outlined in Figure 2. Two systems S_1 and S_2 are assumed. S_1 is generating the products P_1 and P_2, whereas S_2 only produces P_1. S_1 and S_2 are not comparable within LCA, because they fulfil different functions $F_1 = P_1 + P_2$ and $F_2 = P_1$. By expanding system S_2 with an alternative production process of P_2, an equal function can be defined: $F_1 = F_2 = P_1 + P_2$.

In this study, the two processes relevant for system expansion are:

- **Supply of nutrients (N, P, K)**: the conventional wastewater system is expanded by the industrial production and application of the equivalent amount of N/P/K-fertilizers that is saved by nutrient recycling via urine and faeces within SCST systems. If the investigated SCST systems supply different amounts of fertilizing equivalents, SCST systems are also expanded with industrial fertilizer production, so that all systems in comparison deliver the same amount of fertilizer (see chapter 4.1.1).

- **Supply of electrical/thermal energy**: both CS and SCST systems may produce additional electrical or thermal energy (e.g. via biogas from digestion). If additionally produced electrical and/or thermal energy can actually be used, the systems are expanded by conventional production of energy (in power plants or heating plants) in order to establish functional equivalency between all analysed systems (see chapter 4.1.2).

The results of this LCA study are strongly influenced by the secondary functions of the SCST systems, especially by the supply of fertilizer via nutrient recycling. The industrial production of fertilizer is associated with considerable demand for energy and raw materials. Industrially processed fertilizer may also contain elevated levels of potentially harmful substances (e.g. heavy metals in P fertilizers). Hence, the substitution of industrial fertilizer by recycled nutrients from urine or faeces can offer ecological advantages.

However, the fertilizing equivalents of SCST fertilizers may not resemble exactly those of industrial fertilizers in terms of nutrient availability, plant uptake etc. The equivalence between industrial and SCST fertilizers has been investigated in agricultural field tests to determine the substitution potential of SCST fertilizers. Within the KWB research project (SCST, 2006), pot and field tests were carried out with urine and composted faeces as potential substitutes for industrial nitrogen fertilizer. The results and further details about the substitution potential of secondary fertilizers are given in chapter 3.3.4 (see also annex).

2.1.5 Description of the systems

Five different sanitation concepts are compared within this study: a reference scenario and four new sanitation concepts (see Table 4). The reference system is defined as the drainage of mixed domestic waste water and the subsequent treatment in a conventional activated sludge plant, complemented with separate collection and composting or incineration of domestic biowaste. To differentiate the influence of the system expansion processes, a second reference scenario is considered (WWTP) which excludes the system expansion processes, thus displaying the environmental impacts of the wastewater treatment processes alone (without fertilizer and additional energy supply).

All four of the new sanitation concepts include urine separation toilets, where the urine is separated at the source, stored and used as fertilizer. Two of them use gravity toilets for faeces drainage and subsequent composting of faeces, the other two use vacuum toilets for faeces drainage and subsequent digestion of faeces. Furthermore, two alternatives for greywater treatment are investigated: a natural biological treatment in a constructed wetland or a technical treatment process (Sequencing Batch Reactor or SBR). All five scenarios are supplemented with processes of system expansion if necessary. A detailed description of each of the five systems under investigation is provided in the following chapters.

Apart from direct application of separated urine, it can be processed with different treatment technologies to improve handling and eliminate potentially harmful micropollutants (e.g. pharmaceuticals). To determine the ecologically preferable treatment technology, a separate

Table 4: Overview of investigated sanitation systems

Scenario	Urine	Faeces	Greywater	Biowaste	System expansion
WWTP(*)	Conventional wastewater treatment			Composting (partial incineration)	--
REF					Fertilizer + energy
Comp_Nat	Separation → fertilizer	Gravity toilets + composting	Constructed wetlands	Composting (partial incineration)	Fertilizer + energy
Comp_Tech			SBR		
Vac_Nat	Separation → fertilizer	Vacuum toilets + digestion	Constructed wetlands	Digestion (partial incineration)	Fertilizer + energy
Vac_Tech			SBR		

(*) scenario WWTP is not directly comparable to the others due to missing system expansion. It is investigated to identify the share of system expansion processes on the environmental impacts of the reference system

LCA of various urine treatment options is performed. The scenarios for urine treatment are described in chapter 2.1.5.6.

All SCST systems discharge the treated effluent into surface waters. Wastewater reclamation and reuse is not considered within this study, even though treated greywater has a high potential for different reuse options (e.g. flush water, irrigation (Lens et al., 2001)). However, water scarcity is not a problem in Germany, as natural water resources are predominantly sufficient to meet local demands. Legal restrictions on the reuse of wastewater in the household pose another serious barrier for the implementation of concepts for water recycling.

Surface waters in Germany can be highly influenced by wastewater discharge. In consequence, discharge limits for wastewater treatment plants are strict in terms of biological or chemical oxygen demand (BSB_5, COD) or nutrients (N_{inorg}, P_{total}). For this study, the effluent quality should meet at least the legal standards of the current EU regulations.

2.1.5.1 Ref: conventional wastewater treatment

For basis of comparison a conventional system for drainage and treatment of sewage is regarded (Figure 3). The system considers only domestic wastewater without storm water, which is treated separately.

Within the system the drainage and treatment of domestic waste water is considered as well as the collection and composting of organic waste from kitchen, garden and municipal loppings, including the supply of energy, auxiliary material and transports. Sewage is treated by an activated sludge process, which is equipped with a chemical precipitation stage for phosphorus removal. The excess sludge is stabilised by anaerobic digestion. The generated biogas can be used for production of electricity and thermal energy. After stabilisation the sludge is thickened and co-incinerated within a municipal waste incineration plant. The application of stabilised sewage sludge in agriculture, which would constitute a form of nutrient recycling, is not considered within this study. Due to the considerable amount of inorganic and organic pollutants contained in sewage sludge, a discussion about stricter limits for heavy metals and organic pollutants is ongoing, and it tends to prohibit the agricultural utilisation of most of the sewage sludge in Germany in the future.

Typically, a wastewater treatment plant with this relatively small dimension (ca. 5000 inhabitant equivalents) is neither equipped with an anaerobic digester for sludge stabilisation due to economic reasons, nor with a phosphorus removal stage as no nutrient discharge limits are applied for this dimension of treatment plant. However, these assumptions reflect a certain “best-case” of conventional wastewater treatment technology as possible today. Thus, the new sanitation concepts compete against an optimized conventional system, which will probably be operated in this mode in the near future due to stricter effluent limits and energy recovery demands.

The reference scenario is expanded by processes for fertilizer production and energy supply (see 2.1.3 for details).

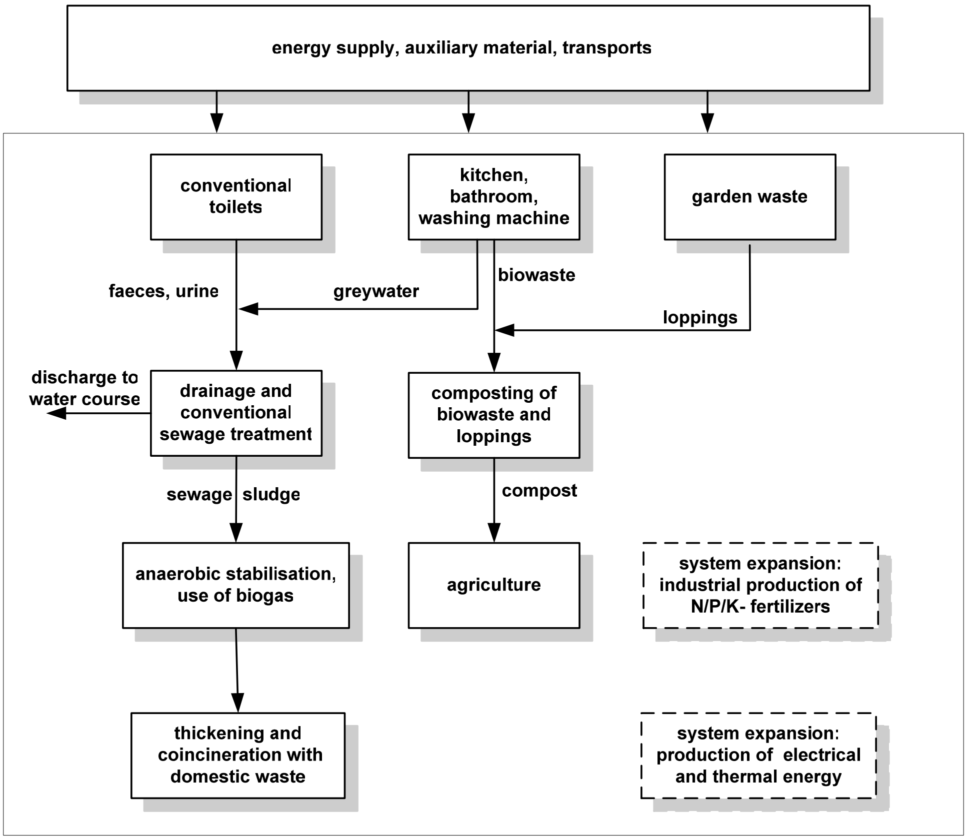


Figure 3: Conventional sanitation system as reference scenario (REF)

2.1.5.2 Comp_Nat: urine separation, composting of faeces and natural greywater treatment

This scenario considers the use of gravity separation toilets, which allow the separate collection of undiluted urine apart from faeces. The separated urine is collected in urine pipes, discharged by gravity to pump wells, from which it is pumped to large holding tanks. Tank trucks carry the urine to surrounding farms, where it is interim stored for at least 0.5 years for sufficient hygienisation before its application on farmland as a secondary fertilizer.

Faeces together with flush water and a small proportion of misled urine are drained off by gravity to pumping stations, from where the mix is pumped to solid-liquid separators. The solid fraction is further thickened and transported by truck to a composting plant, where it is treated in a two-step aerobic composting process together with kitchen waste and loppings. The filtrate from the faeces dewatering is treated together with the remaining household wastewater from kitchen, washing machine, and personal hygiene ("greywater"). The greywater is collected by gravity drainage and treated by sedimentation and biological treatment in constructed wetlands before discharge into surface waters. Sludge from the primary sedimentation is dewatered and incinerated, while the reed from constructed wetlands is added to the composting process. The whole system setup is presented in Figure 4.

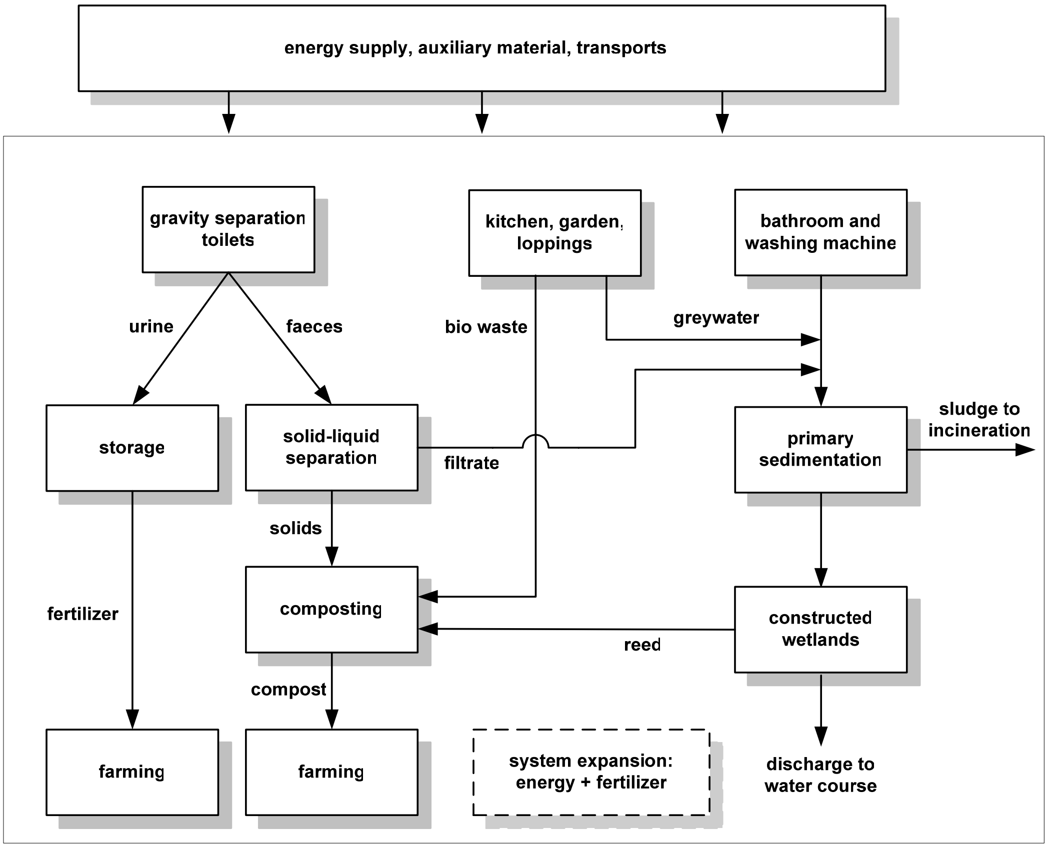


Figure 4: System setup of scenario Comp_Nat

2.1.5.3 Comp_Tech: urine separation, composting of faeces and technical greywater treatment

This scenario is similar to scenario Comp_Nat (see 2.1.5.2). Gravity separation toilets are installed, and urine separation and faeces treatment are designed as described above. Instead of the constructed wetland, greywater and faeces filtrate are treated in a technical wastewater treatment plant. For reasons of symmetry to the reference system, a sequencing batch reactor (SBR) was chosen as the adequate treatment technique. Hereby, sludge from the SBR plant is dewatered and co-incinerated with domestic household waste. A preliminary anaerobic stabilisation with energetic use of biogas (see reference scenario) is not considered due to the smaller volume of greywater sludge and the unknown sludge properties (e.g. organic content, biogas yield). Most likely, digestion of greywater sludge would only yield a small surplus of energy and is therefore neglected. It is assumed that the technical greywater treatment plant stabilises the excess sludge by extended aeration (aerobic stabilisation).

The setup of scenario Comp_Tech is presented in Figure 5.

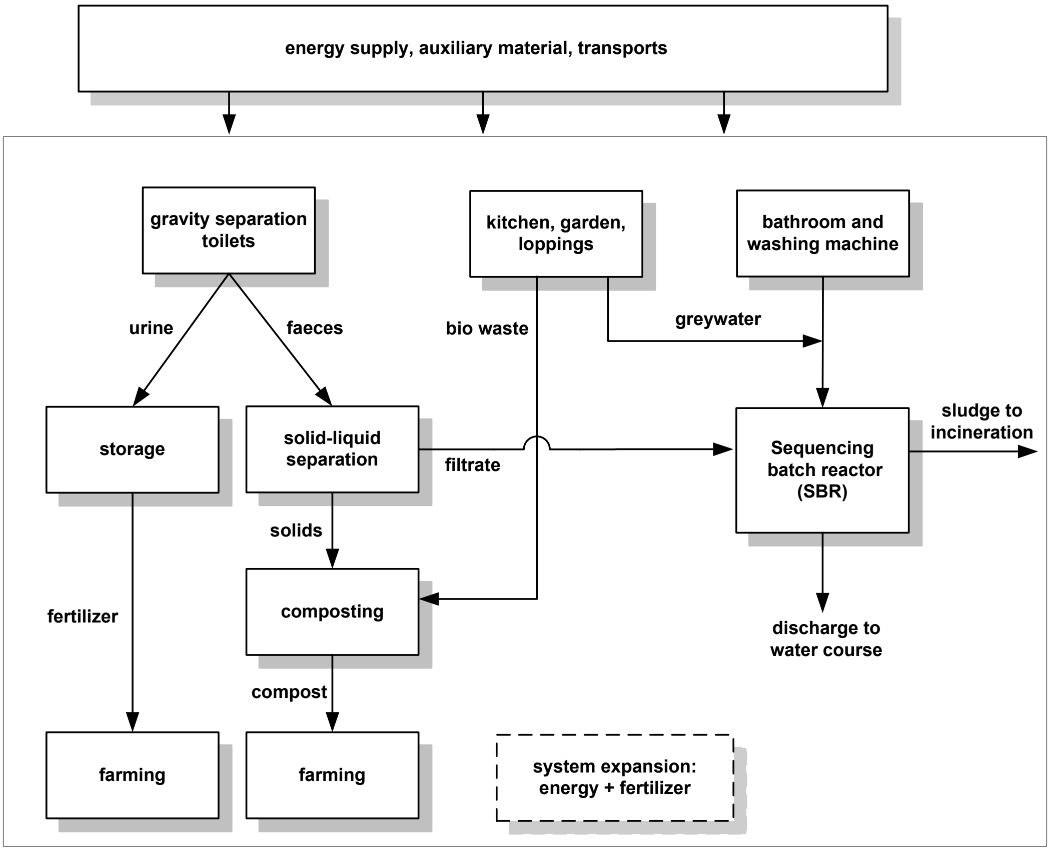


Figure 5: System setup of scenario Comp_Tech

2.1.5.4 Vac_Nat: urine separation, digestion of faeces and natural greywater treatment

In scenario Vac_Nat, vacuum separation toilets are installed in the households. Urine separation and application as secondary fertilizer is done as described above. Faeces are drained by a vacuum system to reduce flush water demand and obtain a mixture with relatively high total solids content. Thus, faeces and flush water can be directly fed into a digestor together with biowaste from kitchen and loppings. In a mesophile digestion process, the organic content is partially converted into biogas, which is used for energy production. The remaining fermentation residue is dewatered and further composted for stabilisation before application to farmland as an organic fertilizer. Wastewater from the dewatering process is treated biologically in a technical unit (SBR) before discharge.

Greywater from households is drained by gravity and treated in a constructed wetland. The reed from constructed wetlands is added into the digestor. Figure 6 depicts the setup of this scenario.

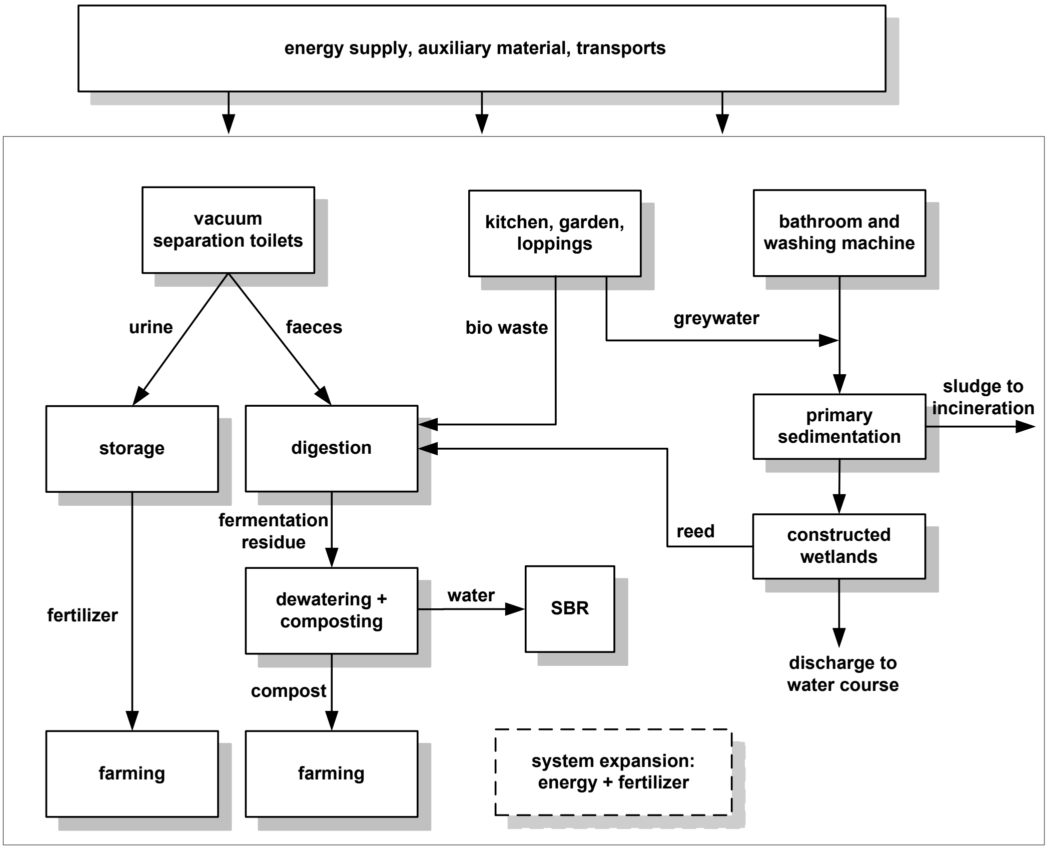


Figure 6: System setup of scenario Vac_Nat

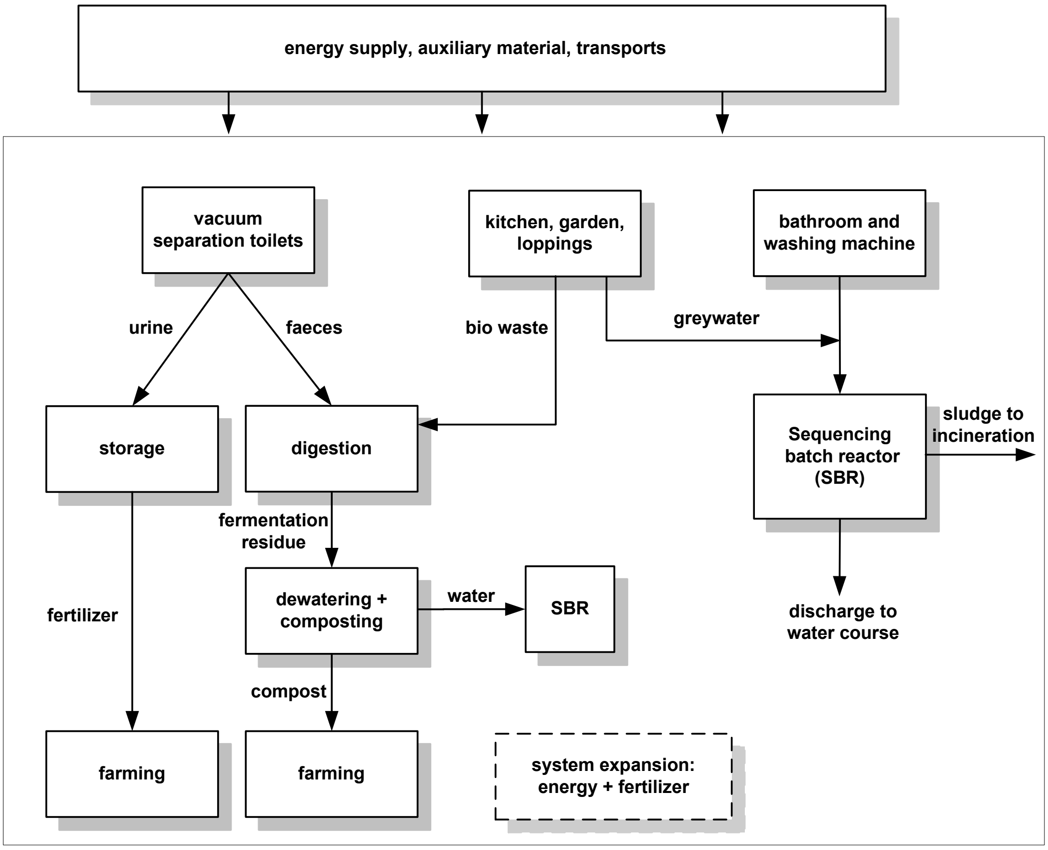


Figure 7: System setup of scenario Vac_Tech

2.1.5.5 Vac_Tech: urine separation, digestion of faeces and technical greywater treatment

This scenario is similar to scenario Vac_Nat. In parallel to scenarios Comp_Nat and Comp_Tech, Vac_Nat and Vac_Tech only differ in the method of greywater treatment. As sanitation facilities, vacuum separation toilets are installed, and urine separation and application and faeces treatment by anaerobic digestion are handled in the same way as in scenario Vac_Nat. Greywater is treated by an SBR plant, and the excess sludge is dewatered and co-incinerated with domestic waste (see Figure 7).

2.1.5.6 Urine treatment options

Urine separation and use as a secondary fertilizer are an essential part of the alternative sanitation systems investigated in this study. The collection and direct application of urine on farmland has been studied in pilot projects in Sweden (Johansson et al., 2001) and, in smaller scale, in Germany (Otterpohl et al., 2001). The environmental impacts of urine separating systems have also been assessed based on the Swedish data (Jönsson, 2002). In all studies, urine was directly applied to the farmland after appropriate storage time for hygienic stabilisation. However, the relatively small nutrient concentration in urine (compared to industrial fertilizer) leads to large volumes that have to be handled and carried to the field. Despite the impractical handling aspects for the farmers, the transport of large volumes of urine requires considerable energy in form of fossil fuels and may offset the energetic advantages of industrial fertilizer substitution. Therefore, a volume reduction or concentration of the nutrients contained in urine is desirable from both practical and energetic aspects.

Beside the valuable nutrients, urine can contain considerable amounts of anthropogenic trace organic substances like pharmaceuticals or hormones (Escher et al., 2006). The potential effects and risks of applying these micro-pollutants on food crops are not properly investigated. Though, the public acceptance of urine as a secondary fertilizer could be improved by precautionary measures for inactivation of organic micro-pollutants to avoid a potential harmful influence on humans or the environment.

Both targets for urine treatment – volume reduction and the elimination of micro-pollutants – can be reached by a number of different techniques. Maurer compared the energy demand of several nitrogen recovery processes (Maurer et al., 2003) and reviewed possible treatment targets and related processes (Maurer et al., 2006). In the present study, three different process combinations for the treatment of separated urine are compared by life cycle assessment with the storage and direct application of urine (Figure 8). Data for process modelling is mainly taken from the review of Maurer (Maurer et al., 2006) and pilot studies in Sweden (Ganrot, 2005) and Germany (SCST, 2006) (see Table 5).

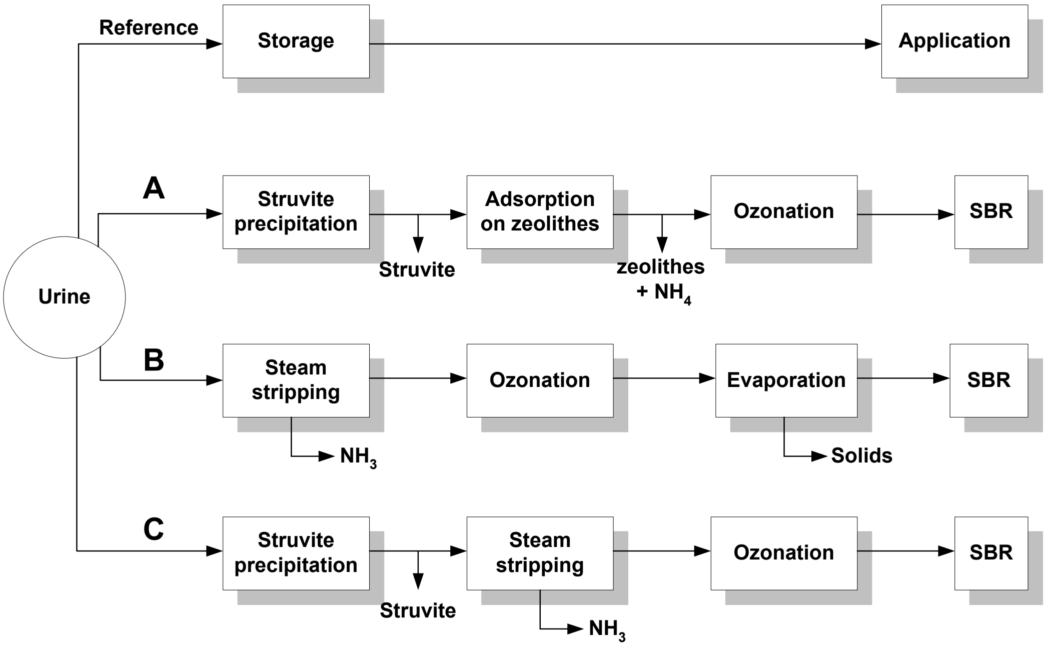


Figure 8: Scenarios for urine treatment

The first process includes precipitation (as magnesium-ammonium-phosphate, MAP or struvite) and subsequent ion exchange with zeolithes for the quantitative recovery of both nitrogen and phosphorus as a solid fertilizer product. Micropollutants are thus separated from the nutrients and remain in the depleted urine, where they are oxidised by ozone before further treatment in an SBR unit.

The second process is based on physical phenomena: steam stripping for the removal of nitrogen in the form of ammonia, followed by ozonation to transform and inactivate the micro-pollutants and subsequent evaporation to recover the remaining nitrogen together with phosphorus and other trace elements (K, Ca). Evaporated water, which contains a small fraction of nitrogen and volatile organic carbon, is treated in an SBR unit.

The third process is a combination: struvite precipitation and subsequent steam stripping. The depleted urine is oxidised by ozonation to transform persistent micropollutants, and then treated in an SBR unit.

Table 5: Comparison of considered urine treatment processes

Process combinations	N recovery [%]	P recovery [%]	Volume reduction	Micro-pollutants	Source
Storage	99	100	-	-	1
Struvite precipitation + Ion exchange	90	99	++	+	2
Steam stripping + Ozone + Evaporation	96	100	+	++	3,4
Struvite precipitation + Steam stripping	95	90	+	++	2,3,4

(1) Johansson et al., 2001 (2) Ganrot, 2005 (3) TUHH: SCST, 2006 (4) Maurer et al., 2006

2.1.6 System boundaries

In order to make a specific statement about the object of interest (i.e. the sanitary systems) and to keep the LCA manageable, system boundaries must be defined that run [Guinée et al., 1993]:

- between the analysed economic system and the environment
- between the analysed system and other economic systems (allocation problem)
- between relevant and not relevant life cycle phases and unit processes
- between relevant and not relevant substance and energy flows
- between considered and not considered geographical regions (local, regional, intercontinental, global) and
- between considered and not considered time periods (period of production, life time of products, time horizon of emissions, etc.).

In the following, important definitions with regard to the system boundaries are made.

2.1.6.1 System boundaries between the analysed economic system and the environment

Normally it is obvious where the economic system ends and the environment begins: Emissions from factories or engines, etc. to air, water and soil pass the system boundary to the environment. However, processes in agriculture or landfill sites have characteristics that apply to ecosystems as well as to economic systems [Guinée et al., 2002].

In this study, agricultural soil is regarded as a part of the environmental system. Nutrients, other fertilizer ingredients and trace components applied to the agricultural soil are treated as emissions into the environment. The incorporation of nutrients and other substances into crops is not specified, because there is no relevance concerning the interests of this LCA study. However, the percentage of the nutrient availability to the crop is considered with regard to the amount of fertilizers that can be substituted by urine or other recycled fertilisers. Secondary emissions by soil erosion or migration of nitrate into the groundwater are not specified in the inventory. These effects are difficult to quantify in a general way and are not included within the impact assessment. Gaseous nitrogen emissions (NH_3, N_2O) during fertilizer application are included in the inventory.

The landfill deposition of slag and ashes originating from co-incineration of biowaste or sewage sludge is not included within this study. Usually, landfill processes are considered as a part of the economic system. However, the environmental impacts of gaseous or leachate emissions from deposition of inert ashes and slag are estimated to be negligible for the result of this LCA. Moreover, a major fraction of these incineration residues is reused today as filling material, e.g. in road construction.

2.1.6.2 Multi function processes and recycling

Economic systems often imply processes that generate several products or fulfil more than one function. This particularly applies in the following cases:

- Combined production of co-products
- Combined waste air, waste water and solid waste treatment as well as combined services like transports within different economic systems
- Reuse and recycling processes

In such cases the associated input and output flows must be allocated between the functions of interest and other functions on the base of physical or economical relationships. Because allocation is often regarded as subjective, ISO 14041 recommends avoiding allocation, for example by dividing the respective unit processes in suitable sub processes, by modelling the processes according to causal relationships or by including additional functions. Latter is done by expanding the system with alternative production routes like it is described in chapter 2.1.3 with regard to nutrient and energy recycling.

The processes of system expansion implemented in this study are listed in Table 6.

Table 6: Products delivered by new sanitary concepts and their respective equivalent conventional products

Additional products delivered by SCST systems	Equivalent products considered in system expansion
Compost out of faeces and biowaste Residuals from digestion used as fertilizers Stabilised urine	Fertilizing equivalents of industrial fertilizers, regarding nutrients N, P, K
Electric energy produced out of biogas in combined power and heat units	Electric energy, produced by power plant mix of Germany
Thermal energy produced out of biogas in combined power and heat units	Thermal energy, produced by conventional heat plants

With regard to the substitution potential of recycled fertilizers, special attention has to be paid to the availability of nutrients. The organic carbon content of the secondary fertilizers (“humus”) is not accounted for in system expansion. Organic fertilizer is supposed to be applied mainly as supplementary fertilizer, and its fertilizing effect cannot be quantified precisely. Thus, it is not clear whether the organic content of secondary fertilizers will really substitute other organic fertilizers (e.g. peat or bark mulch). This study is therefore limited to the recycling of the nutrients nitrogen, phosphorus and potassium (NPK) concerning the equivalence and substitution of industrial fertilizers. This may eventually lead to an underestimation of the substitution potential of secondary fertilizers, but the influence on the overall environmental comparison is supposed to be negligible.

Co-production applies mainly to energy and fuel production as well as to the production of caustic soda and chlorine within the production of auxiliary material. For these purposes pre-allocated data sets form literature and software data base are used, which are widely accepted [IFU and IFEU, 2004; Boustead, 1998; Boustead, 1999]. Generally the allocation is done there on the basis of physical relationships (mass-, energy- molar- proportions).

The combined treatment of solid waste in waste incineration plants and the treatment of sewage in municipal sewage plants modelled widely on the basis of physical and chemical relationships. For waste incineration a model within the software Umberto® is used, where emission, residuals and the demand of subsidiary materials are calculated based on the waste composition and boundary conditions of operation [IFU and IFEU, 2004]. For sewage treatment a model is used that has been developed by the authors especially for the use in LCA studies.

The recycling or disposal of construction material after using phase is considered depending on the type of material (Table 7). Although concrete and vitrified clay are partially recycled after the using phase, no recycling or system expansion is modelled, because concrete needs approximately the same amount of energy to be recycled, as can be later substituted due to the further use of the recycled material [Baitz et al., 2004]. After recycling, concrete is used as substitute for primary gravel, split or sand in ancillary applications. Metallic materials like steel or cast iron are partially recycled in a quasi closed-loop recycling mode, which is already incorporated in the production of these materials (see 3.2.1 for details). Plastic components are assumed to be incinerated for disposal, including the recovery of feedstock energy.

Table 7: Modes of disposal and recycling for construction materials

Material	Mode of disposal
Concrete	Landfill or filling material
Vitrified clay	Landfill or filling material
Steel	Quasi closed-loop recycling (electric arc furnace process) recycling share: 42% (BDSV, 2005)
Cast iron	Quasi closed-loop recycling, recycling share: 88% (BDSV, 2005)
Plastics	Incineration + recovery of feedstock energy

2.1.6.3 Considered life cycle phases and sub-systems

With regard to life cycle phases and sub-systems considered it is distinguished between operational flows and flows associated with the production of capital equipment.

Operation

All unit processes needed for the operation of the analysed systems are included with the exception of processes that are identical in all systems, because these would not result in differences relevant for the environmental evaluation. Generally all unit processes are linked again with all preceding und succeeding processes needed for the production of raw and auxiliary material, energy as well as the treatment and disposal of residuals. In particular the following Life Cycle phases are included:

- Domestic processes: Urination and defecation including flushing, water consumption and pollution by laundry washing and personal hygiene; production of kitchen biowaste and loppings
- Water supply: Pumping and delivery
- Production and supply of electric and thermal energy
- Transports
- Conventional biowaste treatment: Composting and partial incineration
- Conventional wastewater treatment including digestion of sewage sludge and incineration of residuals
- Greywater treatment
- Co-digestion and composting of faeces with biowaste and loppings, application of compost/digestion residual as fertilizer
- Urine storage and application as fertilizer
- Application of conventional fertilizers including nutrient availability

Not included are inspection and repairing expenditures as well as human work and transport of workers: Although this items may be significant, no data is available that is suitable for quantifying environmental related input and output flows.

Finally, drainage and treatment of stormwater and additional percolating water are not considered. Stormwater is supposed to be drained and treated or trickled separately.

Capital equipment

Because of a long lifetime, capital equipment makes often only a minor contribution to the overall environmental impacts and is excluded in many LCA studies. However, taking into account the results of Zimmermann et al. 1996, the environmental expenditures for the construction of sanitary systems may have a significant share of the overall expenditures. In this study capital equipment needed for construction of the sanitary systems is included, as far as the respective components are not identical in each system. Capital equipment of the background system (i.e. production and supply of conventional energy, any industrial production plant, road construction, etc.) is not included.

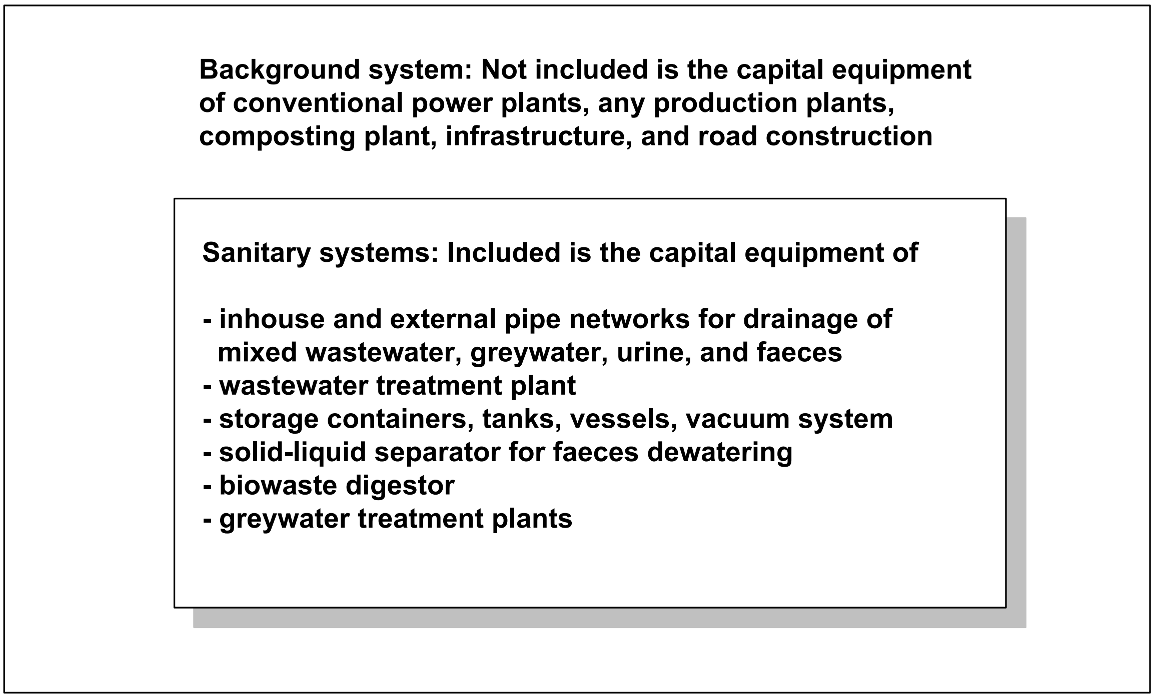
The new sanitary systems are assumed to be integrated in new buildings and a new urban development area. With regard to the construction the following components are considered (see chapter 3.2.3 for details):

- Pipe networks for in-house and external drainage of black and grey water, urine, and faeces, including inspection manholes and pumping wells
- Excavation of trenches
- Conventional wastewater treatment plant
- Containers, tanks and vessels for in-house and external storage
- Pumps, pumping wells and vacuum system parts
- Greywater treatment plants (constructed wetlands or SBR, resp.)
- Solid-liquid separators for faeces dewatering
- Digester and combined power and heat unit

The following components of sanitary systems are not included:

- Production of urinals, toilet bowls: It is assumed that the associated expenses are approximately equal for SCST and CS systems
- Construction of composting plant: Both, CS and SCST systems include composting of biowaste. The scale of the plant depends on the volume to be treated rather than on the mass. As the volume variation of kitchen waste and loppings is much bigger than the additional volume of faeces, the scale of the composting plant within the SCST systems is assumed to be equal to the composting plant of the conventional system.
- Human work and transport of workers
- Energy demand for the installation (except welding energy for HDPE piping)
- Electric measurement and control systems
- Drainage and treatment of stormwater and additional percolating water: This is of particular relevance with regard to the size of the sewer and the sewage plant. In this study stormwater is supposed to be drained and treated and trickled separately.

Figure 9 shows the procedure with regard to the consideration of capital equipment for sanitary systems and the background system.



Background system: Not included is the capital equipment of conventional power plants, any production plants, composting plant, infrastructure, and road construction

Sanitary systems: Included is the capital equipment of

- **inhouse and external pipe networks for drainage of mixed wastewater, greywater, urine, and faeces**
- **wastewater treatment plant**
- **storage containers, tanks, vessels, vacuum system**
- **solid-liquid separator for faeces dewatering**
- **biowaste digestor**
- **greywater treatment plants**

Figure 9: Consideration of capital equipment for sanitary systems and the background system

Cut-off criteria

In order to make the LCA tool manageable and to avoid unlimited datasets, unit processes and subsystems are usually excluded from further analysis, which are considered not to be relevant for the environmental performance of the analysed functions. For this purpose, commonly intermediate product or waste flows are cut off that are smaller than a certain mass or energy share of the total intermediate product or waste input or output (e.g. 3%).

In this study no generally cut-off-criteria is defined. For each process it is decided separately, whether or not an intermediate product (or waste) flow is linked to its preceding or succeeding processes according to its estimated environmental relevance and the availability of data. However, the aggregated share of all substances cut off may not exceed 5% of the total intermediate product input or output mass, respectively energy. With regard to data from literature, cut off criteria mentioned there is used.

2.1.6.4 Considered elementary flows

According to ISO 14041 environmental interventions linked to unit processes are denoted as elementary flows. Elementary flows are for instance resources (e.g. crude oil), emissions to air or land use. In the face of numerous chemical compounds, the number of elementary flows potentially released to the environment is extremely high. Although up to date datasets provide large lists of elementary flows for Life Cycle Inventory, in this study only selected substances are considered with regard to the following criteria:

- elementary flows should be considered in particular that have a known significance in wastewater and biowaste management as well as in agricultural processes
- elementary flows of general environmental interest should be also included
- the selected elementary flows must quantify the considered impact categories in a sufficient way
- the respective elementary flow should be known for each unit process of the system, at least for the most important ones, in order to provide a symmetrical database and to avoid unproved conclusions
- elementary flows that are considered as possibly relevant but are not yet investigated well must be neglected (e.g. organic trace substances like pharmaceuticals in urine)

Table 8 shows the preliminary selected elementary flows. Listed elementary flows are the most relevant for the ecological comparison. In some database models (e.g. for electric energy supply), the inventory lists more than 130 different flows of emissions. However, most of them are not decisive for the LCA result, and are therefore neglected in the further documentation, if not stated otherwise.

The respective impact categories and the method of the impact assessment are documented in chapter 2.2.1. In order to get an impression of the distribution of the single elements within the analysed systems, some material flows are decomposed to their elementary composition (Table 9).

Table 8: Considered relevant elementary flows

Emissions to air	Emissions to water	Emissions to soil	Resources in geol. deposit
CO_2, CO, CH_4	C, N, K, P, S	C, N, K, P, S	Lignite
N_2O, NH_3, NO_x	and respect. species	and respect. species	Hard coal
SO_x			Uranium
HCl, HF	Cd, Cr, Cu, Hg, Ni, Pb,	Cd, Cr, Cu, Hg, Ni, Pb,	Natural gas
	Zn	Zn, Fe, U	Crude oil
PM 10			Phosphorus
NMVOC	Ca, Mg, Na, Cl	Ca, Mg, Na, Cl	Lead
Benzene			Iron ore
Formaldehyde			Copper ore
BaP, PAH			Zinc
PCDD / PCDF (TE)			Bauxite
			Sulphur
As, Be, Pb, Cd, Cr, Co,			Nickel ore
Cu, Ni, Hg, Se, Th, Zn			Potash
			Gypsum

Table 9: Material flows inside the system and their elementary composition

Flows inside the system

Main materials	Components in main materials considered separately
Domestic waste streams: Mixed wastewater, urine, faeces, greywater, biowaste, loppings	C, N, K, P, S Cd, Cr, Cu, Hg, Ni, Pb, Zn Ca, Mg, Na, Cl
Industrial fertilizers	N, K, P, S, Ca,Cl Cd, Cr, Cu, Hg, Ni, Pb, U, Zn
Auxiliary material, e.g.: Precipitation and flocculation chemicals, fuels	Fe, S, Cl
Construction material	No elementary decomposition is considered
Electric, thermal and mechanical energy	

2.1.6.5 Geographical and temporal scope

The reference system refers mainly to average German or Western Europe conditions in the period of 1995 to 2005. The SCST systems represent possible future concepts. However, the considered technology is mainly based upon the same geographical and temporal scope. If no up-to-date data is available, older data has to be used and is identified accordingly.

The international origin of some materials and resources (crude oil, phosphate rock, etc.) is taken into account by local production conditions and respective transport distances.

According to the nature of LCA, no spatial or temporal differentiation of environmental interventions is done. Environmental interventions are aggregated without accounting for time and location. With regard to impact assessment, the method applied in this study mostly assumes an infinite time horizon, with the exception of the global warming potential, where a time horizon of 100 years is considered [Guinée et al., 2002]. The method is described in detail below.

2.2 Life Cycle Impact Assessment

The procedure of Life Cycle Impact Assessment (LCIA) aims to connect, to the extent possible, emissions and extractions of Life Cycle inventories (LCI) on the basis of impact pathways to their potential environmental damages. Impact pathways consist of linked environmental processes, and they express the causal chain of subsequent effects originating from an emission or extraction.

According to ISO 14042, 1999, inventory results are first classified into impact categories. A category indicator, representing the amount of impact potential (e.g. indicator is global warming potential for the impact category of climate change), can be located at any place between the inventory results and the category endpoint. Two main methods have been developed (Figure 10):

A) Classical LCIA methods that stop quantitative modelling before the end of the impact pathways and link LCI results to so-defined midpoint categories

B) Damage-oriented methods which aim at LCA outcomes that are more easily interpretable fur further weighting, by modelling the cause-effect chain up to the environmental damage (damage to human health, ecosystem quality etc.)

For the LCIA in this study, a well-established midpoint-oriented approach is used that was developed at the Centrum voor Milieukunde at Leiden University (CML method, Guinée et al., 2002). The environmental impact of the sanitation systems can be compared regarding each impact category for itself. The results are more transparent and comprehensible.

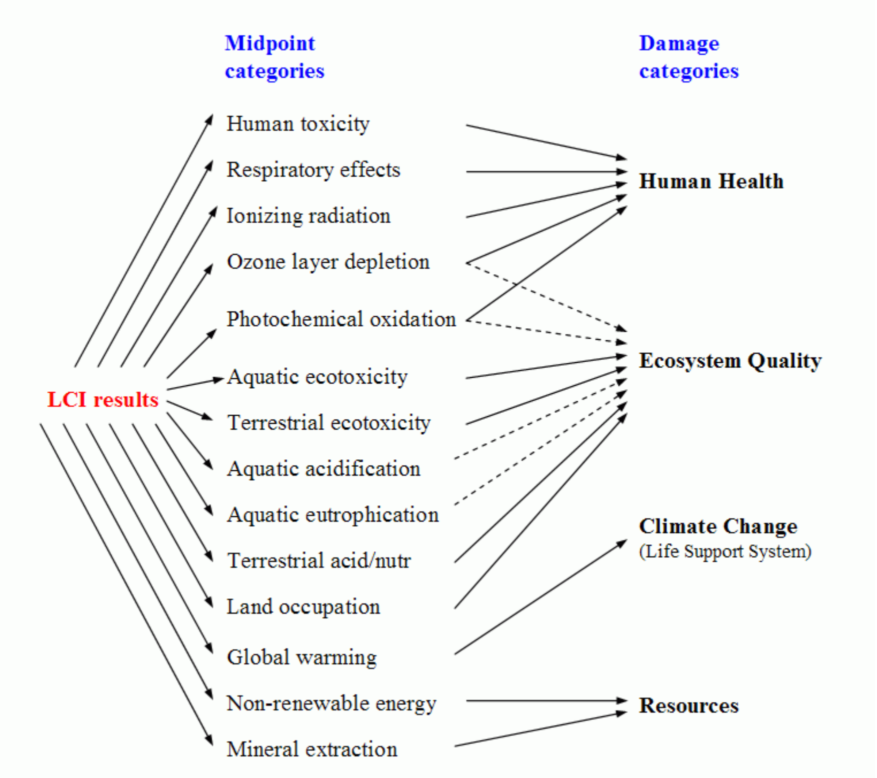


Figure 10: General structure of LCIA methods (adapted from Jolliet et al., 2003)

A damage-oriented approach would produce results which are easier to compare as there are only four damage categories, but the second step of impact modelling (from midpoint to damage categories) would lead to higher uncertainties of the results. Furthermore, the evaluation of differences in damage categories between the compared sanitation systems is more difficult, as the cause of the difference is not necessarily traceable.

However, the LCIA procedure includes further steps for the evaluation of environmental impacts. In general, LCIA consists of the following general steps (ISO 14042, 1999, see Figure 11):

1. **Classification and characterization:** The classification and characterization step is done according to the procedure described above. Extractions and emissions are classified into certain impact categories and characterized by scientifically derived factors. Aggregation of all relevant flows multiplied with their characterization factors results in an overall category indicator.

2. **Normalization:** In order to gain a better understanding of the relative size and importance of a category indicator, a normalization step is required. Each indicator calculated by LCIA is benchmarked against the known total effect for this indicator.

3. **Evaluation and weighting:** Finally, all normalized indicators are aggregated to an overall eco-profile to allow for a simple comparison between the systems, with all indicators regarded as equally important. If appropriate, normalized indicators can be multiplied beforehand by weighting factors to represent the relative importance of the effect. Sensitivity analysis shows the relative sensitivity of LCIA results against the variation of relevant parameters or assumptions in life cycle inventory modelling. Thus, the reliability of the LCIA results and its conclusions can be tested, and important system parameters can be identified.

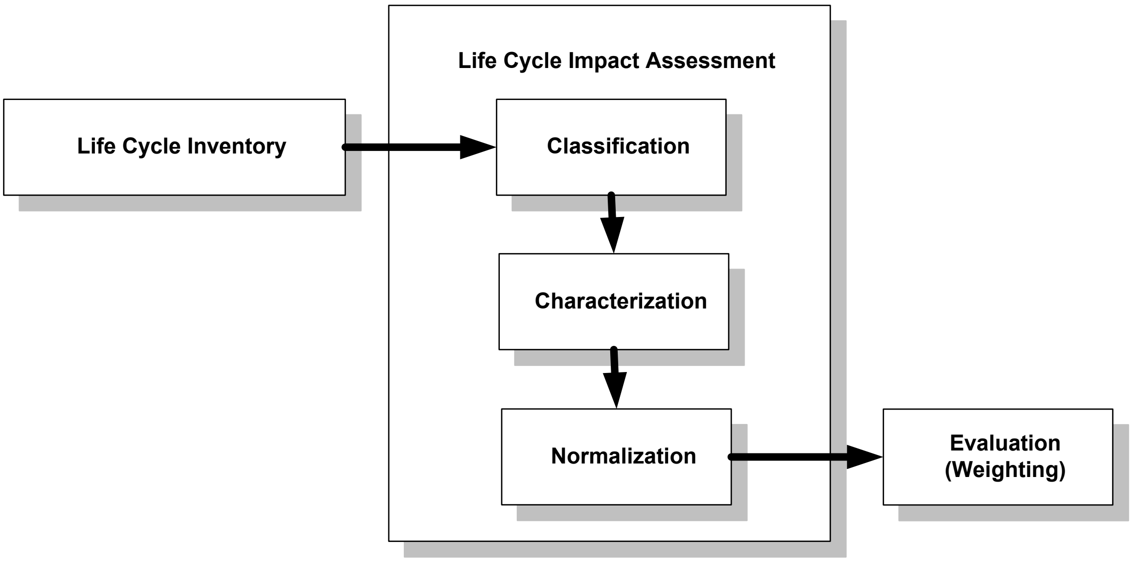


Figure 11: Elements of a Life Cycle Impact Assessment (according to ISO 14042, 1999)

In the following chapters, the impact categories relevant for this study are listed and further discussed. The data for normalization is presented along with some fundamentals for interpretation of the LCIA results and the concluding sensitivity analysis.

2.2.1 Impact categories

The categories of environmental impacts included in the CML method and relevant for this LCA study are listed in Table 10 together with the respective emissions or extractions which contribute to the impact categories. The CML method is complemented with a basic indicator for the overall energy demand as defined in VDI, 1997 (CED).

Table 10: Classification: Considered impact categories and their related emissions and extractions

Impact category	Indicator	Relevant emissions/extractions	Source
		(R: resources, W: emission in water, A: emission in air S: emission in soil)	
Energy demand	Cumulated energy demand (CED)	(R): all forms of primary energy	**1**
Depletion of abiotic resources	Abiotic depletion potential (ADP)	(R): lignite, hard coal, natural gas, crude oil, uranium, raw phosphate, lead, iron ore, copper ore, nickel ore, chromium ore, zinc, bauxite, sulphur, potash	**2**
Climate change	Global warming potential (GWP)	(A): CO_2, CO, CH_4, N_2O	**2**
Acidification	Acidification potential (AP)	(A): NH_3, NO_x, SO_2, HCl, HF	**2,3**
Eutrophication	Eutrophication potential (EP-limP)	(W): PO_4, P species, COD, TOC, (N species)	**2,4**
Aquatic Ecotoxicity	Freshwater aquatic ecotoxicity potential (FAETP)	(W): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn, F (A): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn, HF, benzene, formaldehyde, BaP, PAH, PCDD/PCDF (S): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn	**2**
Terrestrial Ecotoxity	Terrestrial ecotoxicity potential (TETP)		
Human toxicity	Human toxicity potential (HTP)	(W): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn, F (A): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn, NH_3, NO_x, SO_2, HCl, HF, particles, PM10, benzene, formaldehyde, BaP, PAH, PCDD/PCDF (S): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn	**2**

(1) VDI, 1997 (2) Guinée et al., 2002 (3) Huijbregts, 1999 (4) Jolliet et al., 2003

The cumulated energy demand (CED) is a measure for the use of primary energy for a certain product or service, including all relevant pre-chains such as the production of raw materials, transports etc. All forms of primary energy are considered, non-renewable energy sources like fossil fuels or uranium as well as solar or wind power. An index usually indicates if a certain form of CED is meant (e.g. CED_{fossil} for fossil fuels only). The indicator summarizes the energy demand for a product or service and is based on large databases where CED has been calculated for a multitude of materials, electric power generation processes, modes of transport etc. CED is relatively easy to determine and hence seen as a simple indicator to assess the overall environmental impacts related to energy demand.

The abiotic resource depletion potential (ADP) describes the consumption of natural resources which are regarded as non-living. Fossil resources (e.g. lignite, hard coal, crude oil) are included in this category as well as mineral resources such as iron or copper ore, raw phosphates and bauxite.

The global warming potential (GWP) is a well-known indicator for the greenhouse effect caused by the emission of climate-active gases (e.g. CO_2, CH_4, N_2O) into the atmosphere. Climate change is a global phenomenon and poses serious risks to many regions in the world. The characterization factors are taken from the Intergovernmental Panel on Climate Change (IPCC) and describe the potential for global warming equivalent to CO_2 as a reference. The factors are related to a time horizon of 100 years (GWP_{100a}). CO_2 from renewable sources (e.g. biogas from faeces digestion combusted in a central heating plant) does not increase global warming and is therefore not taken into account as being relevant for climate change.

The acidification potential (AP) characterizes the acidifying effect of a number of atmospheric gases (e.g. NH_3, NO_x, SO_2). Through oxidation and hydrolysis, these acids can be deposited as dust (dry deposition) or dissolved in precipitation (wet deposition) and may cause acidification of surface waters and soil. In the CML evaluation method, characterization factors for gaseous emissions are standardized and not related to the region where they are emitted. In this LCIA, updated characterization factors are applied which include fate and transport of air pollutants, considering region-specific factors for Germany and a critical load model (only above threshold changes are counted) (Huijbregts, 1999).

The eutrophication potential (EP) describes the negative effects of excess supply of nutrients to surface waters. The aquatic emission of nitrogen and phosphorus and biodegradable organics (as biological oxygen demand (BOD)) contributes to phytoplankton growth and subsequent oxygen deficiency in rivers and lakes or the marine environment. Nitrogen (as NH_3 or NO_x) may deposit on the water surface and further increase the eutrophication process. However, only one of the two nutrients will normally be limiting in a given water body, typically phosphorus in fresh waters and nitrogen in marine water. Therefore, the standard EP factors from CML deliver a certain overestimation of the eutrophication potential, as both nutrients are counted for. For this study, it is assumed that the emissions take place into a P-limited watershed (i.e. a river or lake, referring to LCIA method IMPACT 2002+ (Jolliet et al., 2003)). Therefore, airborne or aquatic nitrogen emissions do not contribute to EP-limP in this study. However, both cases (with and without nitrogen contribution) will be discussed in the evaluation, and direct emissions of N and P in surface waters will be displayed separately to allow for an extended evaluation of all nutrient emissions.

While the categories of CED, ADP, GWP, AP and EP-limP are relatively well established and the related characterization factors are based on a sound scientific basis, the characterization of eco- and human toxicity (freshwater aquatic ecotoxicity potential (FAETP), terrestrial ecotoxicity potential (TETP), human toxicity potential (HTP)) is far more complex. The resulting toxic effect of emissions into the environment is dependent on the fate of the substance, the exposure time (acute or chronic toxicity) and pathway, and potential cumulative effects together with other toxic emissions. Therefore, the evaluation of toxicity potentials is connected with high uncertainties, with a possible variation up to a factor of 100 for single characterization factors. As a consequence, LCIA methods for the evaluation of toxicity use different model assumptions and characterization factors, and the resulting indicator values can differ over a wide range. In this study, the toxicity factors of the CML evaluation method are used as a basic approach. To check the influence of this evaluation method on the LCA results, a recently developed toxicity evaluation model is applied in parallel (IMPACT 2002+, Jolliet et al., 2003). The respective indicators of IMPACT 2002+ are aquatic ecotoxicity (AET) and terrestrial ecotoxicity (TET). Further details can be found in chapter 4.3.1.6.

Characterization factors for all indicators included in this LCIA are listed for each emission or resource in the annex.

2.2.2 Normalization

According to ISO 14042, 1999, normalization is an essential step for LCA. The results for the different impact categories are divided by their respective normalization factors for better understanding of the relative importance and magnitude of these results for each scenario under study. In the present study, the LCIA results are normalized in relation to the inhabitant equivalents (i.e. the respective value of the indicator per inhabitant and year, e.g. 1000 kg CO_2-eq/pe*a). The following equation demonstrates the procedure of normalization:

$$I_{i,norm} = \frac{I_{i,system}}{(I_{i,Germany} / pop_{Germany})}$$

Equation 1

with $I_{i,\,norm}$ *= normalized value of indicator I [pe*a]*
$I_{i,\,system}$ *= value of indicator i for the assessed system [kg or MJ/a]*
$I_{i,\,Germany}$ *= total value of indicator i for Germany [kg or MJ/a]*
$pop_{Germany}$ *= population of Germany [pe]*

After normalization of the indicators, they can be summed up to produce an ecological profile of the investigated system. Thus, a comparison of the overall ecological impacts is possible to support comprehensive statements about the ecological ranking of the considered system scenarios.

Normalization data is usually included in the LCIA method, and CML gives normalization data for different geographical and temporal scopes (NL 1997, Western Europe 1995, World 1995). However, the present study is located in the German context in terms of technology, legislation etc, and is intended to assess the impacts of sanitary systems to be installed in the near future. Normalization should be carried out in relation to the latest available data for Germany.

Therefore, normalization data for German conditions is collected and implemented in this study (see annex for details). Table 11 shows the generated normalization data for Germany 2002 in comparison to data included in CML method. Whereas for certain indicators all relevant emission data could be collected with reasonable effort, other indicators are limited to accessible German emission data. Resulting inconsistencies in comparison to CML data are accepted for the advantage of specific normalization data for this study. To check the influence of normalization data on the results, CML data for Western Europe 1995 is applied in addition to generated German data.

Table 11: Normalization data

Indicator	Source	**Germany 2002**	**NL 1997**	**Western Europe 1995**	**World 1995**
		1	2	2	2
Cumulated energy demand (fossil)	[GJ/(pe*a)]	146	-	-	-
Cumulated energy demand (nuclear)	[GJ/(pe*a)]	22	-	-	-
Abiotic resource depletion	[kg Sb-eq/(pe*a)]	32,6 (*)	110	32,6	27,7
Global warming potential	[kg CO_2-eq/(pe*a)]	12279	16100	14600	6830
Acidification potential**	[kg SO_2-eq/(pe*a)]	35,4	42,9	84,2	52,9
Acidification potential** (regional + threshold)	[kg SO_2-eq/(pe*a)]	13,5	-	-	-
Eutrophication potential	[kg PO_4-eq/(pe*a)]	9,7	32,1	38,4	22,8
Eutrophication potential (P-limited)	[kg PO_4-eq/(pe*a)]	1,3	-	-	-
Freshwater aquatic ecotoxicity potential	[kg 1,4 DCB-eq/(pe*a)]	88,9	483	1550	359
Terrestrial ecotoxicity potential	[kg 1,4 DCB-eq /(pe*a)]	70,1	61,5	146	47,1
Human toxicity potential	[kg 1,4 DCB-eq /(pe*a)]	7266	12100	23300	8800

(1) see annex (2) Guinée et al., 2002
* ADP adopted from W. Europe 1995 due to lack of data
** this study only uses acidification potential with regional fate factors and threshold concentration effect

2.2.3 Interpretation

This last step of LCIA brings together the results of classification, characterization and normalization to an overall interpretation and consequential conclusions for the ecological assessment. Apart from normalization, other weighting procedures may be applied here to the results of the impact categories. Methodical limitations and the results of sensitivity analysis (see 2.2.4) have to be considered in the evaluation of the LCA results. Data quality (see 2.3.1) and the influence of made assumptions (see 2.3.2) are to be taken into account while interpreting the numerical results of LCA.

In this study, the interpretation of LCA results consider the results of inventory analysis, the single LCIA impact categories, and the normalized LCIA results summed up in eco-profiles. Through sensitivity analysis, key LCA parameters of sanitation systems are identified and presented for future assessments. Final conclusions are drawn with regard to limitations in methodology and data quality of the study.

2.2.4 Sensitivity analysis

Sensitivity analysis is a systematic procedure for the estimation of the influence of selected data or assumptions on the LCA results and conclusions. Specific parameters (e.g. transport distances, system efficiency) are varied over a defined range to observe the resulting effect on certain LCIA impact categories or the overall LCA results. Thereby, the robustness of the LCA results and the consequential conclusions can be described in relation to the data quality. For this LCA study, specific parameters are selected for sensitivity analysis that deal with certain questions arousing during the course of the project. Important parameters with high influence on the decisive indicators of the system comparison are investigated, and relevant recommendations for process design are deducted from the results.

A systematic sensitivity analysis for the majority of parameters would go beyond the scope of this study. However, important key parameters for future LCA studies of alternative sanitation systems are identified from the experiences collected during the work on this LCA study. Together with other valuable experiences, they should make the development and evaluation of LCI models of sanitation systems less time-consuming in the future.

2.3 General remarks

The present LCA study is designed to assess the environmental impacts of different sanitation systems. Modelling the construction and operation of these systems requires an extensive collection and review of data. To keep the LCA manageable, there is always a compromise between the level of detail and data quality on the one hand and the effort and time required on the other hand. Moreover, some processes and features of alternative systems are not properly investigated yet, so projections into the near future have to be made.

The following chapters give a short overview about the data quality, necessary assumptions and methodological limitations of this LCA study. Detailed information about these issues is also given within the inventory analysis (see chapter 3).

2.3.1 Requirements to data and data quality

In general, the quality and significance of LCA results highly depends on the quality of data that is used for the inventory analysis. At best, the inventory is based on consistent measurement data from an existing system which resembles the analysed scenario in system layout, dimension, and location and time of operation (e.g. a conventional sanitation system for 5,000 inhabitants in Germany in the year 2006). However, this optimum data quality cannot be reached in most cases due to lack of adequate time and financial means for data collection. There is a trade-off between the data quality and the required time and effort for the study. Frequently, Life Cycle Inventories are based on data from pilot studies, databases or literature, which have to be readjusted (e.g. upscaling) to fit into the LCA boundary conditions. Where no appropriate data is available, reasonable assumptions have to be made to allow for a coherent inventory analysis. Thus, the assessment of data quality in an LCA is an important indicator for its representativeness and hence for the quality of its conclusions.

In the present study, data is collected from several sources (see Table 12). Data for the supply of construction materials, energy and transport expenditures are mainly taken from databases and supplemented with literature data. Process data for sanitation systems is based on literature and pilot studies. Whereas the conventional system has been studied extensively, alternative systems have been realized and investigated only in pilot scale (e.g. Berlin-Stahnsdorf (subtask 4 of SCST project), Lübeck-Flintenbreite, Freiburg-Vauban). Results from these pilot studies have to be adapted to the conditions of this LCA. For some processes, adequate and consistent data is not available yet (e.g. composting of faeces together with biowaste). In those cases, comparable data from other studies has to be used.

Data quality of this study can be described as medium, given the precondition that the SCST systems under investigation have not yet been realised in full-scale (prospective LCA). Most processes of the alternative sanitation systems have been tested in pilot-scale applications, despite the composting of faeces together with biowaste.

Table 12: Overview of data sources and quality

Process/Flow	Main data sources	Data origin	Reference period	Data quality	Data relevance*
Basic datasets					
Energy					
electrical	ESU-ETH (energy mix: D 2003)	database	1990-95 (2003)	Medium	Medium
thermal	ESU-ETH	database	1990-1995	Medium	Low
Materials					
plastics	APME	database	1995	Medium	Low
concrete/limestone	ESU-ETH	database	1995	Medium	Low
metals	based on ESU-ETH, partially updated	database literature	1990-2000	Medium	Low
flocculant	manufacturer information and assumptions	literature	2000	Medium	Low
Transport by truck	IFU and IFEU, 2004	database	1996	Medium to poor	Medium
Industrial fertilizer production	Patyk and Reinhardt, 1997 Gaillard et al., 1997 Boysen, 1992 (market shares: 1998/99)	literature	1990-1995 1992 1992 1998	Medium	High
Composition of input flow	assumed average values from various publications	literature	1995-2000	Good	High
Construction inventory	exemplary system layout	consulting engineers	2006	Medium to good	Medium
Processes					
General operational parameters	SCST, 2006 Otterwasser, 2005 qualified assumptions	pilot plants literature	2003-06	Medium to good	High
Activated sludge plant	LCA model by TU Berlin data from literature and national surveys	literature	2003	Good	High
Soil filter	SCST, 2006 Otterwasser, 2005	pilot plants literature	2003-06	Medium to good	High
Incineration	IFU and IFEU, 2004	database	1990-2000	Medium	Low
Composting	Vogt et al., 2002	LCA study	2000	Medium to poor	High
Digestion	Vogt et al., 2002 Otterwasser, 2005	LCA study pilot plant	2000 2003	Medium	Medium to high
Urine separation	Stockholm Vatten, 2000 SCST, 2006	review pilot plant	1995-2000 2006	Medium to good	High
Urine treatment	Ganrot, 2005 SCST, 2006	pilot studies	2003-06	Medium	Medium
Fertilizer application	Stockholm Vatten, 2000 SCST, 2006	pilot studies	1995-2000 2006	Medium to good	High

* for the overall ecological comparison: HIGH: impacts on decisive indicator
MEDIUM: strong impacts on non-decisive indicator
LOW: small impacts on non-decisive indicator

2.3.2 Assumptions and limitations

In the following some general and specific limitations of the study are listed.

General limitations

- LCA considers only loads of emissions, without quantifying the concentration of pollutants and without differentiation of time and location of the emission. Particularly with regard to the evaluation of toxic impacts on humans and the environment this is a crucial limitation.
- Some future environmental interventions cannot be quantified by expert knowledge (e.g. final storage of radioactive waste) and must be neglected.
- Sum parameters (NMVOC, COD, etc.) allow only a restricted evaluation of the ecological significance of emissions

Specific limitations

- The results are strongly influenced by the system expansion with conventional production and supply of fertilizers. However, the datasets of these processes are of limited quality as they refer to the beginning of the 1990 decade [Patyk and Reinhardt, 1997]. The heavy metal content of the fertilisers lies between wide ranges, so that averaging is problematic [Boysen, 1992; Vogt et al., 2002].
- Only a limited set of environmental interventions and impact categories are included (e.g. no odour exposure) throughout the analysed systems, because of lack of data and missing symmetry of data
- The effects of organic carbon as a valuable component of compost (e.g. for soil fertility, prevention of acidification processes etc) are not quantified in this study and are neglected in terms of fertilizer substitution (via system expansion). This constraint may lead to an underestimation of the fertilizer substitution potential of alternative sanitation systems.
- Potentially harmful trace organic substances (pharmaceuticals etc.) in human excreta are not evaluated due to missing data of occurrence and toxicity
- No workplace emissions or pathogens are assessed. Such emissions may be particularly associated with the treatment of biowaste.
- Accidents, the risk of accidents or undesirable system behaviour are ignored
- The migration of nitrate into the groundwater and associated impacts are not considered
- The constructional design of waste water systems, the operational expenses and the associated environmental impacts depend on local conditions. Emissions linked with fertilising also depend strongly on several variable boundary conditions (atmospheric conditions, dilution of fertilisers, technique and agricultural machines used, etc.).

3 Life Cycle Inventory (LCI)

The following section describes in detail the life cycle inventories for the construction and operation of the considered sanitation systems. The basic modules for the supply of auxiliary or construction materials and energy are listed below. The present report covers the sources of the data sets, and not the detailed inventories of the database modules. For a complete inventory of a certain basic process, the reader may consult the original sources.

The construction phase is inventoried with detailed information given in the appendices. The operational phase, being the most relevant in this LCA study, is described in detail afterwards. System expansion processes (fertilizer and energy supply) and urine treatment processes are characterized in the last part of this section.

3.1 Basic modules

3.1.1 Energy Supply

The expenditures for the supply of electrical energy are calculated by database models provided by the Umberto® software (IFU and IFEU, 2004). They are mainly based on data from GEMIS database (Fritsche, 2001).

Whereas the considered power plant technology originates mainly from the period of 1990-2000, the average power plant mix was updated with data from Germany in 2003 (BMWA, 2005). Table 13 summarizes the respective data together with the heating values of the different fuels. The process chain includes all steps of energy supply, including the extraction and transport of fuels and the energy transport from power plant to customer. It is assumed that all electricity is drawn at medium voltage, so that the energy losses from transport and transforming add up to 1.8% of the energy production (Frischknecht et al., 1996).

Table 13: Power mix/efficiencies of electricity supply in Germany 2003 and respective heating values

Power plant type	**Proportion in power mix** (1)	**Energy efficiency** (2)	**Fuel**	**Heating value**
	[%]	[%]		[MJ/kg]
Hard coal	23.7	37.2	Hard coal	29.2
Lignite	28.1	30.1	Lignite	21.1
Nuclear	33.0	31.0	Uranium	6375
Oil	1.3	41.7	Crude oil	40.6
Gas	9.3	35.7	Natural gas(3)	45.0
Hydro	2.9	100.0	--	--

(1) BMWA, 2005; missing to 100%: wind, solar, waste incineration
(2) electrical energy output to thermal input
(3) density 0.776 kg/m³

The supply of thermal energy is also based on Umberto® database and GEMIS data. The respective power mix and degree of thermal efficiency is listed in Table 14. The power mix for thermal energy production is heavily depending on the particular situation and location of interest. Thermal energy production is frequently coupled with electrical energy production, as most power plants produce both types of energy. For this study, thermal energy is produced in single heating plants, and an average fuel mix is assumed.

Table 14: Power mix and efficiencies of thermal energy supply

Power plant type	Proportion in power mix	Thermal efficiency
	[%]	[%]
Hard coal	20	85
Lignite	10	85
Light fuel oil	20	85
Heavy fuel oil	10	85
Gas	40	90

3.1.2 Transport by truck

The transport processes within this study are described with the respective model from Umberto® (IFU and IFEU, 2004). This model calculates fuel consumption and emissions relative to the transport distance, the weight of the goods, the utilisation ratio, the type of truck and the modes of traffic. For this study, only the transport distance and the weight of the transported goods vary, whereas the remaining parameters are assumed to be constant for each transport process. The utilisation ratio is set to 100% for one way and 0% for the return trip (average utilisation ratio: 50%). The truck type is set to a single truck (> 20 tons), and the distribution of traffic modes is assumed to 37% on highways, 41% on country roads, and 22% in the city. Table 15 lists all transport processes of this study and the respective distances. The distances are estimated from literature and own assumptions. As the transport of large volumes (e.g. urine) needs a considerable amount of energy, the effect of variation of the transport distance is investigated in sensitivity analysis.

One exemption is the collection and transport of biowaste. Biowaste from households and municipal greens is collected by truck in a stop-and-go mode, before it is transported to a composting plant. The Umberto® database provides a module for biowaste collection and transport, which is applied in this study. The average distance for biowaste collection in stop-and-go mode is set to 7.5 km.

Table 15: Transport processes and distances

Load	From	To	Distance
Fertilizers			[km]
Mineral fertilizer	Manufacturer	Farmer	100
Compost	Collection	Compost plant	15
	Compost plant	Farmer	20
Faeces	Separator	Compost plant	15
Digester residue	Biogas plant	Farmer	20
Urine	Holding tanks	Farmer	10
Waste			
Sludge	WWTP	Incineration	30
Biowaste	Household	Collection	(*) 7.5
Biowaste	Collection	Incineration	30
Auxiliary			
Chemicals	Manufacturer	WWTP	300
Zeolithes	Manufacturer	Urine treatment	1000
Construction materials			
Concrete	Manufacturer	Settlement	50
	Settlement	Disposal	50
Others	Manufacturer	Settlement	300
	Settlement	Disposal	100

(*) stop-and-go collection of biowaste from households

3.1.3 Incineration plant

The co-incineration of biowaste or sewage sludge in an incineration plant for domestic wastes is described by an Umberto® module for a municipal solid waste incineration plant (IFU and IFEU, 2004). It represents an average of the German plants in operation today (grate firing, four-step flue gas cleaning). The module calculates the substantial and energetic expenditures, the emissions and residues, and the electrical or thermal energy derived from the incineration of the waste. In case of a heating value of less than 1,85 MJ/kg waste or a dry matter content of less than 30%, extra fuel is required to maintain the incineration process. The default parameters for the energy output are set to 10% electricity and 30% effective heat of the net calorific value of the waste input. The heating value of the biowaste or sewage sludge is calculated roughly via Dulong´s formula. The supply of auxiliary material (coke, lime, light fuel oil) required during the plant operation is included within the process.

3.1.4 Auxiliary material

The supply of auxiliary materials is described mainly by database modules from Umberto® software, partly amended with other data for certain processes. Table 16 shows a list of the relevant materials and their data sources. Some of the material modules are taken from an

LCA study of water treatment technologies (Ruhland, 2004). Detailed documentation of these modules is provided there.

Table 16: Auxiliary materials of system operation

Material	Application	Source	Remarks
Polyacrylamide	Coagulation aid	Ruhland, 2004	data from Boustead, 1999
Ferric chloride	Flocculant	Ruhland, 2004	production from waste acid
Coke	Incineration plant	IFU and IFEU, 2004	
Light fuel oil	Incineration plant	IFU and IFEU, 2004	
Lime	Incineration plant	Ruhland, 2004	
Magnesium oxide	Urine treatment	Ruhland, 2004	analogical to lime
Zeolithe	Urine treatment	Ruhland, 2004	analogical to lime (without burning)

3.2 System construction

This LCA study includes both the operation and the construction of the different investigated sanitary systems. Whereas other LCA studies of alternative sanitary systems neglect the construction phase (e.g. Bengtsson et al., 1997), the present study includes the expenditures for system build-up in the environmental comparison. Separating waste flows at the source results in the necessity for multiple pipe networks. These surplus expenditures in construction may offset some of the advantages from the alternative systems (e.g. primary energy demand) and turn the comparison in favour of the conventional systems. To examine the influence of the construction phase on the overall LCA result, the material and energy flows for the construction of the different sanitation systems are determined with a high level of detail.

However, the implementation of alternative systems in the defined dimension of settlement (ca. 5000 inhabitants) has not been realized yet. Pilot plants in smaller dimension (10-150 inhabitants) have been built, but the construction data – if available – cannot be transferred directly to a larger settlement. Thus, construction data has to be generated for the alternative systems. For reasons of symmetry, it was decided that the construction data of the conventional system should be generated with the same methodology, even though respective data for the construction of conventional sewers and treatment would be available.

The settlement structure is taken from an existing quarter of the Berlin area to reflect realistic conditions. For this settlement, a conventional and four alternative sanitation systems are laid out in close cooperation with consulting engineers (Otterwasser GmbH) who participated in the SCST project. Their planning expertise and know-how in designing alternative sanitation systems (e.g. Lübeck-Flintenbreite) provides a sound basis for a realistic estimation of the material demand for system construction.

3.2.1 Materials

The components of a sanitary system are manufactured of different materials. Whereas sanitary installations (in-house piping) are mainly made of cast iron or plastic, drainage pipes for the sewer system are made of plastic or vitrified clay. Manholes and pumping shafts are made of concrete, and facilities for treatment (e.g. activated sludge plant, soil filter etc) can include various materials. Table 17 summarizes the construction materials and the respective LCI data sets for their production. The transport distance for building materials from supplier to the settlement is assumed to be 300 km (concrete: 50 km).

Table 17: Construction materials and their LCI data sources

Material	Components	Source	Remarks
Pipes			
Polypropylene	Sanitary pipes, urine pipes	1	based on Boustead, 1999
Polyethylene	Sewage pipes, vacuum pipes	1	based on Boustead, 1999, incl. welding energy
Cast iron	Sanitary pipes	1,3	Recycling share: 88%
Vitrified clay	Sewage pipes	1,3	incl. sealing
Other			
Concrete	Manholes, basins, buildings, digester	2	
Construction steel	Buildings, basins	2	Recycling share: 42%
Stainless steel	Machinery, vacuum plant	2	Recycling share: 42%
Cast iron	Manhole covers, pumps	1,3	Recycling share: 88%
Copper	WWTP, SBR	2	
Aluminium	WWTP, SBR	1	
Polypropylene	Urine pumping shafts	1	mouldings
Polyethylene	WWTP, SBR, soil filter	1	mouldings
Glass fibre reinforced plastic	Urine tanks	1	epoxy resin and glass (1:1)
Limestone	WWTP, SBR	1	
Excavation	Trenches, basins	1	3.83 MJ E_{mech}/m^3 soil

(1) IFU and IFEU, 2004
(2) Ruhland, 2004
(3) Jeschar et al., 1995; Jeschar et al., 1996;

Most of the inventories for material supply are delivered with the UMBERTO® software. They are based on approved databases and are not described in detail in this report. Other material modules are characterized in detail in Ruhland, 2004. For some piping materials (cast iron, vitrified clay), process data from other sources is connected to basic modules of UMBERTO® software.

Beside the materials, another important factor of the construction expenditures is the excavation of trenches and the connected energy demand in form of fossil fuel for machinery. The demand of mechanical energy for excavation is estimated to be 3.83 MJ/m³ soil. The emissions of the excavator operation are included in the inventory. However, the additionally required excavation for alternative systems is minimized by laying the multiple pipes together in one trench wherever possible.

3.2.2 Settlement structure

The settlement structure is adopted from an existing quarter of Berlin with ca. 5000 inhabitants. The quarter (Berlin-Nikolassee) is located in the south-west of the Berlin city area and has a sub-urban character. The distribution of house types and other data of the settlement structure are taken from GIS (Geographic-Information-System) data of the Berlin water works (Table 18). The quarter is divided into a northern part (3627 inhabitants) and a southern part (1264 inhabitants). A map of the respective area is provided in the annex.

The design of the drainage and treatment systems is based on the real distribution of inhabitants throughout the area. For simplification of sewage dimensioning and slope, it was assumed that the whole terrain is even.

Table 18: Settlement structure of the study area

Parameter	Value	Unit	Remarks
Population	4891	inhabitants	
Area	126	ha	
Population density	39	inhabitants/ha	
Buildings	1000		public and commercial buildings excluded
One-family houses	649	units	870 house units
Semi-detached house	98	units	
Row houses	123	units	
Apartment houses	130	units	estimated: apartment houses with 9 apartments (3 floors, 3 apartm. per floor)
Apartments	1170	units	

3.2.3 Inventory

The construction inventory of the sanitary systems includes the in-house sanitary installations, the complete drainage system, and the treatment facilities for the different waste flows (mixed wastewater or urine, faeces, and greywater). Although the construction inventory may depend heavily on local conditions, the following calculations allow for a first estimation of the contribution of the construction phase to the overall ecological comparison.

3.2.3.1 Sanitary installations

The realization of the sanitary in-house piping is strongly depending on the particular architecture of the houses and apartments. For reasons of simplicity, a prototype layout for a house unit and an apartment unit is assumed according to the respective legal norm, so that an estimation of the necessary pipe length and diameters is possible. Results of the calculation are presented in Table 19 (for details see annex). Materials for in-house piping are polypropylene for house units and cast iron for apartments (due to fire prevention measures). Urine collection pipes and vacuum pipes are made of polypropylene. The additional weight of bends, tees and plug-in connections is estimated via a proportional factor. Collection tanks with interface valves for the vacuum system are included in the inventory. The sanitary piping includes the base pipe (material: PP) as far as the house shaft. Sanitary equipment (toilets, sinks etc) is excluded from the inventory.

Table 19: Piping for sanitary in-house installations

Material	**Ø** [mm]	**Weight** [kg/m]	**REF** [m]	**COMP** [m]	**VAC** [m]	**Weight factor*** [%]
Polypropylene	50	0.285	12180	15960	15960	15
	70	0.454	0	14385	14385	15
	100	0.938	10875	23475	12600	15
	150	1.943	6300	6300	0	10
Polyethylene	50	0.666	0	0	17070**	10
Cast iron	50	4.3	7020	7020	7020	10
	70	5.9	0	3510	3510	10
	100	8.4	6338	6338	0	10
Total length		**[m/inhabitant]**	**8.7**	**15.7**	**14.4**	
Total amount of plastics		**[Mg]**	**29.2**	**51.5**	**45.1**	
Total amount of cast iron		**[Mg]**	**91.8**	**114.6**	**53.7**	

(*) proportional addition for bends, tees etc (PP: including 5% for plug-in connections)
(**) plus collection tanks for vacuum system: 1260 units (material weight: 5 kg PE and 2.2 kg cast iron)

3.2.3.2 Drainage system

The drainage system includes house shafts, house connections to the main sewer, and the sewer system with piping and inspection chambers, ending at the treatment facilities. The sewer system design follows the common rules in sewer layout. The resulting length of the sewer pipes is presented in Table 20 (for details and a layout map see annex). The pipes of the conventional system have the largest diameter (DN 150 - 400) due to the combined volume of wastewater flows. For the alternative scenarios, the dimensioning of the drainage system is adjusted to the volume of greywater and brown water, and the possible location of the treatment units (solid-liquid separation, soil filter etc). Urine collection comprises of gravity drainage to pumping shafts and pressure pipes. Materials for the pipes are vitrified clay (conventional sewer, brown water) or plastic (greywater, urine, vacuum).

Inspection chambers, pumping shafts and pressure pipes are included in the layout. The total amount of required material for the drainage system is listed in Table 21. The amount of soil material from excavation is calculated from the required depth of the piping due to sufficient slope for gravity drainage. Whereas gravity drainage pipes are divided into sections with different depth levels (2 – 5.5 m), pressure pipes and vacuum system run in a constant frost-resistant depth of 1 m. Greywater, brown water and yellow water systems are laid in one trench wherever possible to minimize the required excavation. Thus, the amount of excavation material does not differ significantly between scenarios.

The length of the drainage systems differ with a factor of 3 between the conventional and the alternative systems (one pipe network for combined sewage or three pipe networks for urine, faeces, and greywater). However, the total length of the sewage system heavily depends on the settlement structure.

Table 20: Length of drainage systems

System	Material	House connections	Drainage		Pressure pipe
			Diameter	Length	
		[m DN 150]	[mm DN]	[m]	[m DN 50/65]
Conventional	Vitrified clay	10000	150 - 400	13805	0
Brownwater	Vitrified clay	10000	150	11108	7816
Greywater	Plastic	10000	150 - 200	13638	0
Urine	Plastic	10000	150	11108	7050
Vacuum	Plastic	10000*	65 - 100	13308	2241

for details see annex
(*) DN 50

Table 21: Total material demand for drainage systems in different scenarios

Material		REF	COMP	VAC
Vitrified clay	[Mg]	725	557	0
Plastic	[Mg]	5	140	151
Concrete	[Mg]	2762	2743	2737
Cast iron	[Mg]	128	148	146
Excavation	[m³]	44260	47530	47410
Total drainage length	**[m/inhab.]**	**4.9**	**16.5**	**15.8**

including pipes, inspection chambers, pumping shafts, and urine tanks
for details see annex

3.2.3.3 Treatment facilities

The construction expenditures for the treatment facilities of the different scenarios are estimated by a schematic layout of the required system parts. Table 22 gives a summary of the system components which are included in this inventory. As precise material data for most of the components is not available (and presumably depends on local conditions), it has to be adopted from other LCA studies or pilot plants. Where no comparable data could be acquired, material data is roughly estimated. In the following, the sources for important data sets are briefly described, whereas detailed information can be found in the annex.

Table 22: System components of treatment facilities

Ref	Comp_Nat	Comp_Tech	Vac_Nat	Vac_Tech
Conventional wastewater treatment plant (incl. digester)	Solid-liquid separation	Solid-liquid separation	Vacuum system	Vacuum system
	--	--	Biogas plant	Biogas plant
	Urine tanks	Urine tanks	Urine tanks	Urine tanks
	Soil filter (+ sedimentation tank)	SBR	Soil filter (+ sedimentation tank)	SBR
	Service buildings	Service buildings	Service buildings	Service buildings
Not included: composting plant				

For the conventional wastewater treatment plant, a detailed data set of the required amount of materials is adopted from another LCA study on sanitation systems (Schneidmadl, 1999). The data originates from an activated sludge plant with digester (21,000 inhabitant equivalents) and is recalculated in proportion to the influent volume. Construction data for SBR plants could not be obtained, so the SBR plant data for technical greywater treatment is estimated from the conventional activated sludge plant via the influent volume. The material data for the soil filter is adopted from material specifications of the soil filter system in Lübeck-Flintenbreite (Oldenburg, 2002) and related to the required surface area. Sedimentation tanks are additionally provided as primary treatment. The number and dimension of the urine tanks is adopted from the system layout of Otterwasser for the cost calculation. Data for the vacuum system is adopted from manufacturer information, and for the biogas plant data is complied from various sources. The equipment for the solid-liquid separation process (in composting scenarios) is not commercially available yet, so the required material is roughly estimated. For each alternative scenario, three small service buildings (70 m^2) are provided to contain the various treatment facilities.

The composting plant is not included within the construction inventory according to the considerations of chapter 2.1.6.3.

In general, the construction inventory for the treatment facilities has the highest uncertainty and the lowest data quality of the entire construction data. Due to the necessary assumptions, the few available data sets, and the dependence on local conditions, the realistic expenditures for construction of treatment facilities may differ considerably. However, the present calculation allows estimating of the contribution of the facilities to the overall construction expenditures and possible differences between the scenarios.

3.2.4 Service life

In order to allow the addition of the LCI data from construction and operation of the investigated sanitary systems, the expenditures for the construction have to be scaled to a time frame. This can be done by relating the material and energy flows of the construction to the estimated service life of the system components. The estimation of an adequate service life of sanitary systems for LCA purposes is a difficult task. Depending on a rather high or low estimation, the influence of the construction on the overall LCA comparison can be manipulated. For economic calculations, the estimated service life for all components of a sanitary system has been assessed (LAWA, 2005). However, these values relate more to an economic amortisation than to a realistic material-dependent service life.

Table 23 lists the estimated service life for the relevant components of the investigated systems in this study. The values are adopted from the cost calculations in the SCST project (SCST, 2006). Hence, they reflect a more economic point of view and are probably smaller than the realistic material lifetimes. On the other hand, this LCA study considers the construction of sanitary systems, but excludes expenditures for both maintenance and repair of pipes, facilities etc. These exclusions are neglecting the effort necessary to reach a long service life for the sanitary systems. The assumption of a shorter service life can roughly offset these limitations. Furthermore, the contribution of the construction phase to the overall ecological comparison appears to be small, and the increase of the maximum service life would further decrease this influence.

Table 23: Estimated service life of sanitary system components

Components	**Service life**	
	LAWA, 2005	**This study**
	[a]	[a]
Drainage		
Sanitary pipes	--	40*
Drainage pipes	50-80	50*
Shafts	50-80	50*
Pumps	8	12.5
Facilities		
Buildings	30-50	40
Tanks, basins	30-40	40
Machinery	8-30	12.5
Vacuum plant	25-40	40
Soil filter	12-15	40

(*) independent of material

3.3 System operation

This section describes the calculation of the operational expenditures and emissions of the different sanitation systems. For each scenario, the different processes for the treatment of urine, faeces, greywater, and biowaste are characterized concerning the demand of energy and resources, the allocation of elemental input flows to the different output flows, and further emissions occurring from these processes.

General operational parameters of the sanitation systems are the consumption of drinking water for flushing purposes (Table 24) and the distribution of the different waste flows to the treatment processes (Table 25). This study assumes the application of water saving measures even for the conventional scenario. With a low-flush toilet (6 L per use, 3 L per small flush), the water consumption for toilet flushing can be limited to 21 L per person and day in the conventional scenario. The same amount of flush water is assumed for the gravity separation toilets of the composting scenarios. The vacuum separation toilets use around 5 L/(pe*d), thus saving 16 L/(pe*d) or more than 78 m³ flush water per day in the settlement. Despite the saving of drinking water as a probably limited resource, energy for drinking water production can be saved as well. For the treatment and supply of drinking water, an energy demand of 0.5 kWh/m³ is assumed.

It has to be noted that flushing water after urination is not collected with the separated urine in alternative sanitation scenarios, but is drained and treated together with faeces. The separation toilets investigated in this study are equipped with a valve control which prevents flush water from entering the urine pipe. Thus, concentrated urine can be collected for fertilizing purposes.

Table 24: Drinking water consumption of sanitation systems

Scenario	Flushing for faeces	Flushing for urine	Greywater	Water consumption
	[L/(pe*d)]	[L/(pe*d)]	[L/(pe*d)]	[L/(pe*d)]
WWTP	6	15	80	101
REF	6	15	80	101
COMP_NAT	6	15	80	101
COMP_TECH	6	15	80	101
VAC_NAT	2.4	2.8	80	85.2
VAC_TECH	2.4	2.8	80	85.2
Expenditures for drinking water production: 0.5 kWh/m³				

While the complete amount of greywater and faeces is collected and treated, the urine separation process is assumed to separate 75% of the daily urine. The separation efficiency of urine separation toilets has been assessed in several pilot studies (Jönsson, 2001). Depending on the motivation and information of the tenants, 60-90% of urine could be effectively separated. The urine which could not be separated is treated together with faeces and flush water.

For the kitchen biowaste, it is assumed that 80% of the total mass flow are collected and subjected to further treatment (composting or digestion), while 20% are incinerated in a municipal waste incineration plant. For the loppings, the respective quota is 50% to composting/digestion and 50% to incineration.

Table 25: Distribution of waste flows

Flow	Distribution
Faeces	100% to treatment
Urine	if urine separation: 75% separation efficiency (25% misdirected urine to faeces treatment)
Greywater	100% to treatment
Biowaste	80% to treatment (composting or digestion) 20% to incineration
Loppings	50% to treatment (composting or digestion) 50% to incineration

3.3.1 Conventional system

The conventional system comprises of gravity drainage of wastewater and subsequent treatment in an activated sludge plant with anaerobic sludge stabilisation. Biogas from sludge digestion is combusted in a central heat and power plant, while stabilised sludge is dewatered and co-incinerated in a municipal waste incineration plant. Biowaste is collected, composted and used as organic fertilizer.

3.3.1.1 Drainage

The drainage of conventional wastewater is done by gravity. The operation of lifting or pumping stations within the sewer system and the associated energy demand is not considered in this study. Even though terrain properties often require lifting or pumping of wastewater (as for example in the Berlin sewer network), gravity drainage per se does not need energy for operation. Possible transformations of wastewater components during drainage (e.g. the formation of H_2S) are also neglected.

3.3.1.2 Wastewater treatment plant

For the operation of the conventional wastewater treatment plant (WWTP), an LCA process model is applied that was developed at TU Berlin. It describes the operation of an activated sludge WWTP and covers the processes of mechanical, biological and chemical treatment of conventional wastewater, including the anaerobic stabilisation of sewage sludge and the production of biogas. The model calculates the allocation of the different elemental components of wastewater (C, N, P etc) to the various output flows of WWTP effluent, air, sewage sludge. It does not give a dynamic modelling of the wastewater treatment process, but allocates the elements based on linear input-output relationships via specific factors. In addition, the demand of energy and chemicals is determined as a function of process parameters and wastewater loads. The model can be adjusted to specific operational conditions with a set of parameters. Some of the important process parameters are listed in Table 26, while a detailed description of the process model is provided in the annex.

Table 26: Process parameters of conventional WWTP

Parameter		Unit	Remarks
Elimination of COD	35	%	particulate fraction, removal via sedimentation
	95	%	total
Elimination of N	11	%	particulate fraction, removal via sedimentation
	80	%	removal of dissolved N
Elimination of P	17	%	particulate fraction, removal via sedimentation
	95	%	removal of dissolved P, effluent: 0.8 mg P/L
removal via Bio-P	20	%	rest: chemical precipitation ($FeCl_3$)
Elimination of heavy metals	60-85	%	depending on element
Sludge age	20	days	
Yield coefficient	0.67	g $C_{biomass}$ per g $C_{substrate}$	
Dry matter content stabilised sludge	40	%	dewatering with centrifuge
Electrical energy	0.63	kWh/m³	total (without energy benefit from biogas use)
	0.62	kWh/kg $COD_{eliminated}$	
	23.1	kWh/(design-pe*a)	
Thermal energy			demand is met by waste heat from CHP plant

The WWTP of the reference scenario is assumed to operate with nitrification and denitrification and chemical phosphorus elimination via simultaneous precipitation. Particulate fractions of COD, N, and P are removed via sedimentation and do not contribute to aeration energy or chemical demand. Heavy metals are bound in element-specific fractions to sewage sludge. The thermal energy demand for digester heating is met by the waste heat from biogas combustion in a central heat and power plant (CHP plant). The electrical energy demand is calculated via the particular energy needs of the processes involved (e.g aeration, mixing, pumping, sludge treatment etc). The resulting total energy demand is in the range of average values for German WWTP (Table 27). For a small plant (ca 5000 inhabitant equivalents), the specific energy demand is considerably low, and the energy balance is further improved by energy production from biogas combustion. Thus, the reference scenario reflects a WWTP technology where the energy balance is optimized towards energy savings and recovery. The elimination of COD, nitrogen and phosphorus is estimated rather high and above the average efficiency of a WWTP of this dimension (there are no discharge limits for nitrogen and phosphorus), which again reflects a certain best-case of conventional wastewater treatment technology for the reference scenario.

Table 27: Energy demand of 1,097 municipal WWTP in Germany (LfU, 1998)

Relation	Median	80%-percentile	weighted average
kWh/(design-pe*a)	27,0	41,0	24,3
kWh/(pe * a)	41,5	64,0	31,7
kWh/m³	0,32	0,56	0,32
kWh/ (kg COD)	1,06	1,7	0,88

The biogas is combusted in a CHP plant (efficiencies: electrical 32%, thermal 57%) with a lean engine (spark ignition, 60kW) with high oxygen excess. An operation-related flaring of 5% of the generated biogas is considered. Details of the CHP plant and emissions from biogas combustion are specified in the annex.

For the wastewater treatment process, exemplary transfer coefficients calculated with the default parameters describe the allocation of the elemental input flows to the different output flows (Table 28). The sludge contains 25% of carbon, 18% of nitrogen, and 96% of phosphorus of the wastewater load. Effluent loads and concentrations are calculated in chapter 4.3.1.5. The stabilised sludge is dewatered to a dry matter content of 40% and transported by truck to a waste incineration plant (30 km), where it is co-incinerated with domestic waste for energy production.

Table 28: Transfer coefficients of elemental flows in WWTP

Input	Emission as	Effluent [%]	Air [%]	Sludge [%]	Biogas [%]
COD-C	COD-C	5.0			
	HCO_3-C	1.8			
	CO_2-C		44.7		8.1
	CH_4-C				15.4
	C_{org} in sludge			25.0	
P_{total}-P	P-species	4.2			
	P in sludge			95.8	
N_{total}-N	NH_4-N	3.6			
	NO_3-N	10.7			
	N-species	3.6			
	N_2-N		64.0		0.1
	NH_3-N		0.3		
	N_2O-N		0.2		
	N in sludge			17.5	

3.3.1.3 Collection and composting of biowaste

The biowaste is collected by truck in a stop-and-go mode (7.5 km) and transported to a composting plant (30 km). The composting process layout is similar to the composting of biowaste together with faeces: pretreatment of biowaste, followed by an encapsulated intensive composting in boxes (11 days) and a subsequent open composting in piles (56 days). Exhaust air from the intensive composting process is cleared in a biofilter.

LCI data for the composting process is adopted from an extensive study of biowaste composting (Vogt et al., 2002). Table 29 summarizes the energy demand of the different processes, and transfer coefficients of the complete composting process are listed in Table 30. Allocation of elemental flows, energetic expenditures, and emissions are calculated as described in detail in chapter 3.3.2.2 (composting of faces and biowaste). Gaseous emissions of nitrogen (NH_3, NO_x, N_2O) and methane are most relevant for the environmental evaluation. Leachate of the intensive composting process is recycled for the moistening of the piles during open composting. Excess leachate is discharged into the sewer and treated in the conventional WWTP.

Table 29: Energy demand for the composting of biowaste (Vogt et al., 2002)

Process	Unit	Energy demand
Pretreatment	[kWh/Mg biowaste]	14.2
Intensive composting	[kWh/Mg biowaste]	3.0
Open composting	[L diesel/Mg biowaste]	0.5

Table 30: Transfer coefficients of elemental flows in two-stage composting of biowaste (Vogt et al., 2002)

Input	Emission as	Compost [%]	Air [%]	Excess leachate [%]
Organic carbon	C_{org} in compost	40.6		
	TOC		1.2	
	CO2-C		55.8	
	CH_4-C		1.8	
	COD-C			0.6
N_{total}-N	N in compost	60.1		
	NH_3-N		32.3	
	N_2O-N		2.3	
	N_2-N		0.8	
	NO_x-N (as NO_2)		4.2	
	N-species			0.3
P_{total}-P	P in compost	100.0		
K	K in compost	100.0		
salts	salts in compost	100.0		
heavy metals	h.m. in compost	100.0		

3.3.2 SCST systems

The alternative sanitation scenarios investigated in this report include different concepts for the treatment of separated flows of urine, faeces, greywater, and biowaste. While urine separation and application as fertilizer is applied in all scenarios, faeces are treated either by aerobic composting or anaerobic digestion. Greywater is treated either with a planted soil filter or an activated sludge plant (sequencing batch reactor). In the following chapters, each part of the alternative systems is characterized in detail in terms of technical realization and the respective Life Cycle Inventory. An overview of all investigated scenarios is provided in chapter 2.1.5.

3.3.2.1 Urine separation

The urine is source-separated with urine-diverting toilets. The efficiency of the separation process is estimated to be 75%, meaning that 75% of the total daily urine flow can be separated and used for fertilizing purposes, whereas the remaining 25% of urine are drained and treated together with faeces and flush water. Experiences from Swedish pilot plants indicate that 60 – 90% of total urine flow can be separated depending on the motivation of the tenants (Jönsson, 2001). The influence of this important system parameter on relevant environmental indicators is further investigated in sensitivity analysis (see 4.4.1).

The separated urine is drained by gravity to collection shafts (max 100 m), from where it is pumped to holding tanks. The tanks are designed for a maximum holding time of 14 days before they have to be emptied. From there, the urine is transported by truck (10 km) to the farms. The urine is stored in large tanks for more than six months, following the recommendations of Swedish studies on hygienisation of stored urine (Stockholm Vatten, 2000). After that storing period, the urine is considered to be hygienically safe for application as fertilizer. Prior to the application, the urine is mixed to stir up possible precipitates and diluted with service water (1:1). The urine handling system is depicted in Figure 12.

For the LCI of urine separation, the energy demand for the various pumping processes is estimated to 0.08 kWh/m^3 urine. The treatment and supply of service water for dilution is assumed to consume 0.1 kWh/m^3.

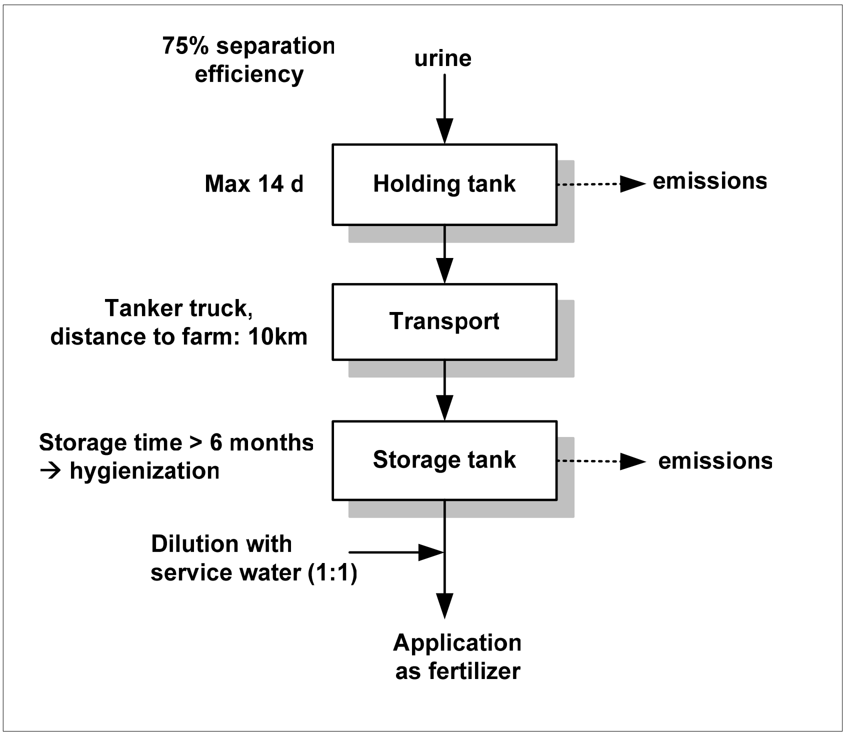


Figure 12: System layout of urine separation

During the collection and storage of urine, the urea content is hydrolysed to ammonia, thus rising pH from 6 to around 9. A fraction of the nitrogen is lost via ammonia emission during collection, transport and storage. Table 31 lists some transfer coefficients from literature. Most of the ammonia evaporates during the application of urine on the fields. Depending on the application technique and weather conditions, the losses of nitrogen via ammonia evaporation can amount to 1-10 % (Palm et al., 2002). Pilot-scale field tests conducted in the KWB project resulted in a nitrogen loss of 6-14% (KWB, 2006). This study assumes an average loss of 6% of the nitrogen through ammonia evaporation, if suitable application techniques are adopted. Losses via N_2O and NO_x are estimated to be equivalent to those of mineral fertilizer application. Indirect environmental effects like the conversion of evaporated ammonia to N_2O are neglected.

Table 31: Nitrogen losses during urine collection, storage and application

Losses	Unit	Piping, holding tanks	Storage tanks	Application	Source
NH3-N	% N Urine	<1 / **0,01** / <0,3	-- / **0,003** / --	<10 / **6** / 1-10	1) / 2) / 3)
N_2O-N	% N Urine			**1,25**	4), 5)
NO_x-N	% N Urine			**0,7**	5)

Bold: this study
1) Stockholm Vatten, 2000
2) Vinneras et al., 1998
3) Palm et al., 2002
4) Tidaker, 2003 (equivalent to mineral fertilizers)
5) EMEP/CORINAIR, 2004 (equivalent to mineral fertilizers)

3.3.2.2 Gravity drainage and composting of faeces with biowaste

The aerobic stabilisation and treatment of faeces in a composting process is applied in two of the investigated scenarios. The concept includes the drainage of faeces by gravity together with flush water, followed by an automated solid-liquid separation process, where the flush water is separated from the solids. Thus, faecal matter with sufficient dry matter content is obtained for a subsequent composting process to transform the human waste into a valuable organic fertilizer. Composting of faeces for the production of organic fertilizer is a well-known treatment for small-scale sanitation units or dry toilets. If operated correctly, it has proven to effectively reduce pathogens and odour problems of faecal matter, producing a stabilised soil conditioner with some nutrients. Laboratory pilot studies have revealed that composting with worms ("vermicomposting") is a promising process to convert faeces into an odourless, earth-like material within three months (Shalabi, 2006). However, the composting of human faeces together with biowaste has not been investigated and tested within the dimension of a larger settlement. Both solid-liquid separation and large-scale composting of faeces may pose difficulties for process engineering. In the KWB project, flush water is separated from faecal matter with filter bags (SCST, 2006), but the separation process is unsatisfactory, and the implementation in large-scale settlements is questionable due to impractical handling.

Due to the lacking data for the composting process, it was decided that the composting model of this LCA study should be based on data from an extensive study on biowaste composting (Vogt et al., 2002). Although the process configuration of biowaste composting may differ significantly from a possible composting process for faeces and biowaste (e.g. vermicomposting), the data provides an insight into the amount of resources and emissions connected to the composting process.

In the following, the specific components of a hypothetical system for the drainage and composting of faeces (Figure 13) are described in detail together with the important data for substance flows, resources usage, and emissions.

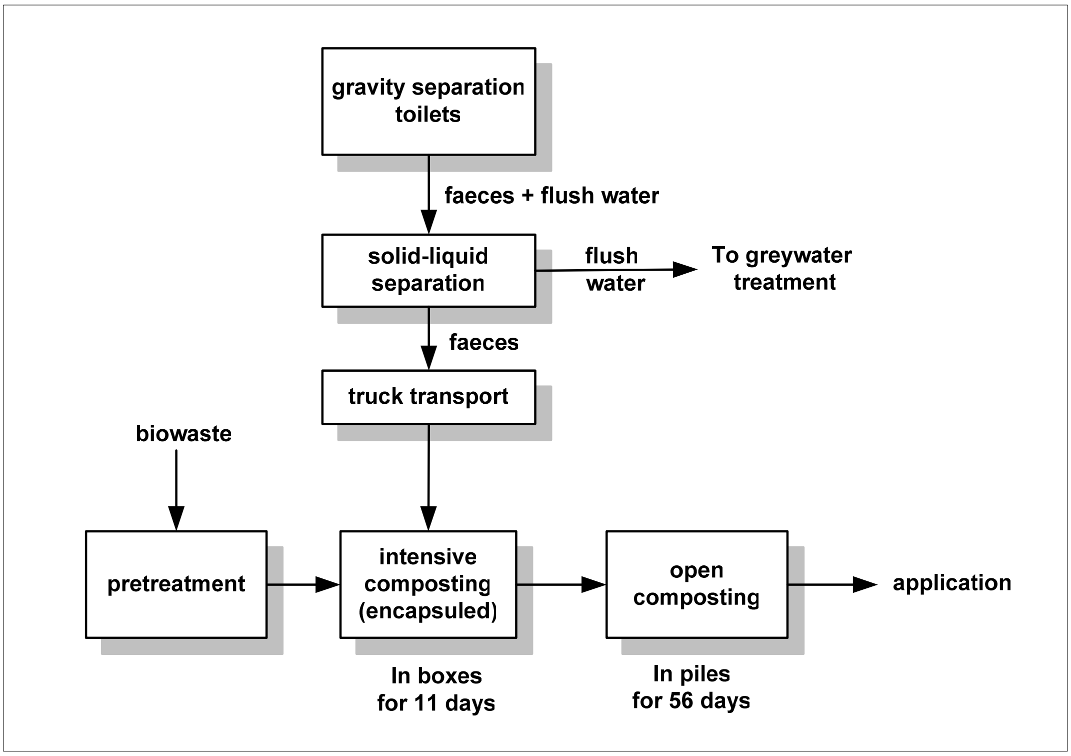


Figure 13: System layout for composting of faeces and biowaste

Solid-liquid separation

The solid-liquid separation process is assumed to be a two-step process with a sedimentation stage followed by a mechanical thickening device (e.g. disc thickener). By adding organic coagulation aids (polyacrylamide), a residual sludge with 10% dry matter content is produced. Beside the demand of electrical energy and chemicals (Table 32), the composition of the faecal filtrate is crucial, because the nutrients contained in the filtrate are lost for recycling purposes and have to be removed by further treatment. Depending on the efficiency of urine separation, the misdirected urine is drained together with the faeces and can lead to elevated concentrations of N and P in the filtrate. The filtrate composition and the respective amounts of “lost” nutrients assumed in this study are listed in Table 33.

Table 32: Energy and chemical demand of solid-liquid separation process

Input	Amount	Remarks
Polyacrylamide	10 g/kg dry matter	analogical to the thickening of sewage sludge (Müller et al., 1994; Schumann et al., 1997)
Electrical energy	0.03 kWh/kg dry matter	
Electrical energy	0.025 kWh/m³	pumping (lifting of faeces)
	0.14 kWh/m³	pumping (filtrate to greywater treatment)
	0.03 kWh/m³	miscellaneous (Sludge pumping etc)

Table 33: Composition of faecal filtrate

Substance	Concentration	Load	Remarks
	[mg/L]	[kg/a]	
Filtrate volume		$37500*10^3$	treatment in greywater unit
Total organic carbon	400	14984	mostly from faeces
Nitrogen (total)	101	3784	= 84% of N from misdirected urine
Phosphorus	12	449	= 100% of P from misdirected urine
Salts, heavy metals	like flush water		

The assumptions are based on measurements of filtrate composition in the pilot plant (SCST, 2006) and consideration of input flows. Whereas 100% of phosphorus and 84% of nitrogen of misdirected urine are lost via the filtrate, organic carbon load of filtrate originates mostly from faeces. The filtrate (ca. 37500 m^3/a) is pumped to the greywater treatment unit. The residual sludge from faecal matter is temporarily stored in dewatering containers and transported by truck to the composting plant.

Pretreatment of biowaste

The biowaste has to be conditioned and homogenised prior to the composting process. Beside the removal of possibly hazardous material (e.g. plastics or metal compounds) for the compost quality, the biowaste with high water content has to be chopped and mixed with structure material (e.g. dry loppings, saw dust etc) if necessary to achieve a sufficient pore volume for maintaining aerobic conditions during the composting process. The additional input of structure material is neglected in the mass balance, as is the further handling of the removed contraries. Table 34 lists the energy demand of the respective steps of pre-treatment. The process is encapsulated and has an exhaust air treatment.

Table 34: Energy demand of pretreatment of biowaste (Vogt et al., 2002)

Process	Energy demand
	[kWh/Mg biowaste]
Exhaust air treatment	8.1
Drum screen	3.0
Magnetic separator	0.5
Shredder	2.6

Intensive composting

The automated process of intensive composting is conducted in closed boxes with heat insulation over a period of 11 days. Table 35 lists the respective allocation coefficients of the different elemental components of biowaste to the output fractions. They have been adopted from a comparative study of biowaste composting (Vogt et al., 2002) and are related to the average biowaste composition. Due to the lack of adequate data for the composting process of faeces and biowaste, the allocation coefficients are adopted par to par. Depending on the process concept and control, allocation coefficients may vary significantly, especially if excessive water content (e.g. from faeces) results in anoxic conditions.

Table 35: Substance flows of intensive composting of pretreated biowaste mixed with loppings (closed boxes, 11 days retention time)

Input biowaste	**Output**	**Source**
(%): exempl. composition, related to input-fresh mass		
Water content + additional water demand (62,5% + 2% of input fresh mass)	In compost: 50% water content	1)
	In biofilter: difference	
	In leachate/condensed water: 125 L/Mg input compost	1)
Organic carbon (12%) Organic dry matter (ca. 23%)	In compost: $C_{out} = (1-0,16)\ C_{in}$	1)
	In biofilter: $CO_2\text{-}C_{out} = 0,140 * C_{in}$ $CH_4\text{-}C_{out} = 0,004 * C_{in}$ $NMVOC\text{-}C_{out} = 0,003 * C_{in}$	1), 3)
	In leachate/condensed water $TOC = 0,013 * C_{in}$	1)
N (0,3%)	In compost: $N_{out} = (1 - 0,106) * N_{in}$	1)
	In biofilter: $NH_3\text{-}N_{out} = 0,096 * N_{in}$ $N_2O\text{-}N_{out} = 0,002 * N_{in}$ $N_2\text{-}N_{out} = 0,002 * N_{in}$	1)
	In leachate/condensed water: $NH_4\text{-}N_{out} = 0,003 * N_{in}$ $N\ org._{out} = 0,003 * N_{in}$	1)
Salts (K, Cl, etc.: ca. 1,7%) and heavy metals	Assumption: salts and heavy metals are accumulating in compost. Fractions lost with leachate are recovered by moistening the raw material with leachate. Fractions lost with excess leachate are neglected.	2), 1)
Misc. inert substances (8,4%)	Remain completely in compost	
Energy demand (incl. biofilter)	10 kWh/Mg waste	1)

1) Vogt et al. 2002
2) Gronauer et al. 1997
3) Leinemann 1998 (similar values from tunnel reactors)

Salts and heavy metals are supposed to accumulate in the compost: The leachate is used to moisten the raw materials prior to the composting process, and hence the salts and heavy metals contained in the leachate are recycled to the compost. Therefore, the output loads with excess leachate are neglected (Gronauer et al., 1997). This assumption has to be proved, but data for heavy metal concentration in leachate implies that the proportion of heavy metals lost with the leachate is small (Fricke, 1990). This study proceeds on the assumption that potential excess leachate is only charged with nutrients and COD. It has to be further treated in a wastewater or greywater unit.

In most composting plants, the exhaust air in cleaned in a biofilter to prevent the output of strong odours and ammonia. However, recent research indicates that biofilters can be a source for climate-active gases like CH_4 and N_2O (Cuhls, 2001; Clemens and Cuhls, 2003). Table 36 shows the allocation coefficients of the biofilter process compiled from different data sources. This study assumes a partial conversion of ammonia into N_2O within the biofilter. Thus, off-gas emissions of N_2O are significantly above the loads specified in Vogt et al., 2002. The actual extent of N_2O emissions depends on the ammonia load to the biofilter. Data for nitrogenous airborne emissions vary considerably in literature, so that the conversion factor of $NH_{3,in}$ to N_2O_{out} is varied in the sensitivity analysis.

Table 36: Allocation coefficients of a biofilter with upstream gas scrubber

Input	Output		Source
H_2O-vapor	Off-gas: H_2O_{out} = H_2O_{in}		
CO_2-C	Off-gas: CO_2-C_{out}	= CO_2-C_{in}	1) 2)
CH_4-C	Off-gas: CH_4-C_{out}	= CH_4-C_{in}	1)
NMVOC-C	Off-gas: NMVOC-C_{out}	= 0,5 * NMVOC-C_{in}	2)
NH_3-N	Off-gas: NH_3-N_{out} N_2O-N_{out1} NO-N_{out}	 = 0,4 * NH_3-N_{in} = 0,6 * 0,26 * NH_3-N_{in} = 0,6 * 0,74 * NH_3-N_{in}	Assumption according to 1) and 3)
N_2O-N	Off-gas: N_2O-N_{out2}	= N_2O-N_{in}	Assumption according to 1), 2) and 3)
N_2-N	compost: N_2-N_{out}	= N_2-N_{in}	2)

1) Clemens und Cuhls 2003
2) Vogt et al. 2002
3) Cuhls 2001

Open composting

For the open composting process following the intensive phase, the required retention time is set to 56 days to reach a stabilised product. A mechanical circulation of the piles with appropriate machinery is necessary biweekly. The allocation coefficients of the open composting (Table 37) are adopted from Vogt et al., 2002.

Table 37: Substance flows of open composting of biowaste (open piles, 56 days retention time)

Input fresh compost	Output	Source
Water content + water demand	Compost (stabilised): 40% water content	1)
	In off-gas: difference	
	Leachate: 0 L/Mg input	1)
Organic carbon	In compost: $C_{out} = (1-0,52)\ C_{in}$	1)
	In off-gas: $CO_2\text{-}C_{out} = 0,494\ C_{in}$ $CH_4\text{-}C_{out} = 0,016\ C_{in}$ $NMVOC\text{-}C_{out} = 0,010\ C_{in}$	1)
Nitrogen	In compost: $N_{out} = (1 – 0,33) * N_{in}$	1)
	In off-gas: $NH_3\text{-}N_{out} = 0,317 * N_{in}$ $N_2O\text{-}N_{out} = 0,007 * N_{in}$ $N_2\text{-}N_{out} = 0,007 * N_{in}$	1)
Salts, Ca, P, etc, heavy metals and misc. inert substances	Assumption: salts and heavy metals accumulate in compost. Open composting has no leachate. Inert substances remain in compost	2), 1)
Diesel fuel (circulation)	0,76 L/Mg waste	1)

1) Vogt et al. 2002
2) Gronauer et al. 1997

3.3.2.3 Vacuum drainage and digestion of faeces with biowaste

Two scenarios of this study use a vacuum system to drain of faeces and flush water, followed by an anaerobic treatment of faeces together with biowaste in a digester. In the fermentation process, the organic content is degraded under anaerobic conditions and transformed into biogas (mainly CO_2 and CH_4). Due to the high heating value of methane (50 kJ/g CH_4), biogas can be used to generate electrical and thermal energy in a central heat and power plant (CHP plant). The remaining residual sludge still contains valuable nutrients and some organic material, making it a good fertilizer for agriculture.

During the anaerobic fermentation process, the organic matter is transformed into small molecules (hydrolysis), converted to acetate, hydrogen and carbon dioxide (acido- and acetogenesis), and finally to methane. Methanogenic bacteria are strictly anaerobic and die off quickly in the presence of oxygen. Hence, the optimum water content lies between 90 and 97%. Digesters can be operated at different temperature ranges: psychrophilic (<20°C), mesophilic (30-42°C), or thermophilic (43-55°C). The required retention time rises with low temperatures. Most of the digesters in Germany are operated in mesophilic conditions and have a retention time of 30-40 days, in which around 50% of the degradable substrate is converted into biogas (BLU, 2004; Stadtmüller, 2004).

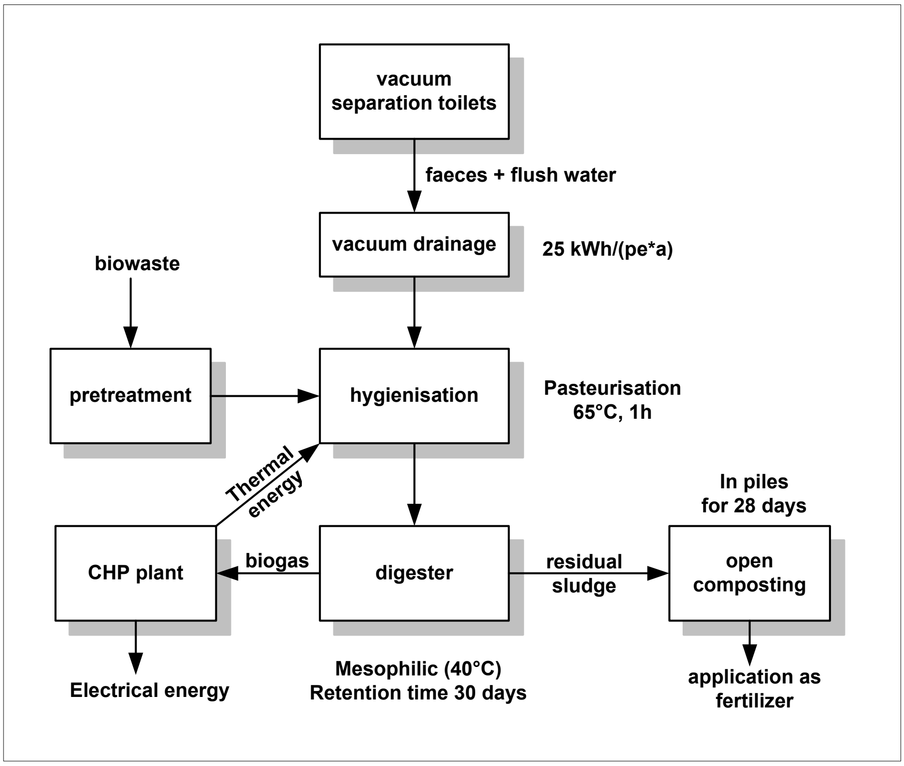


Figure 14: System layout for vacuum drainage and digestion of faeces

Figure 14 shows the different process steps for the vacuum drainage and anaerobic digestion of faeces together with biowaste. The faeces are mixed with pretreated biowaste in a hygienisation stage prior to the digestion process. The digester itself is operated at mesophilic conditions (~ 40°C) with an expected retention time of 30 days. The generated biogas is incinerated in a CHP plant, and the resulting thermal energy is recycled to the hygienisation process. The residual sludge is dewatered and stabilised in an open composting stage before application as a fertilizer. In the following, the substance and energy flows of each process are described in detail.

Vacuum drainage

The vacuum system for the drainage of faeces can operate with small amounts of flush water (1.2 L/use). A vacuum pump imposes a slight negative pressure (ca. - 0.5 bar) on the pipe network, which sucks the mixture of faeces and flush water towards the vacuum tank. Interface valves and small interim holding tanks in the houses allow for comfortable operation without inconvenience for the customer. From the main vacuum tank, faeces are pumped by pressure to the biogas plant. The energy demand for the vacuum and pressure pumps can vary depending on the layout of the system. Table 38 lists energy data from several vacuum systems. For this study, an average energy demand of 25 kWh/(pe*a) for the vacuum pumps is assumed, and another 0.05 kWh/m^3 for the pressure pumps.

Table 38: Energy demand of vacuum systems

System	Amount of wastewater	Inhab.	Energy demand	Remarks	Source
	[L/(pe*d)]		[kWh/(pe*a)]		
Flintenbreite	5	108	51	Not working to capacity	Otterwasser, 2005
Flintenbreite	5	--	17	Possible	Oldenburg, 2002
Vauban	8.4	40	7	Calculated from operating time	Schneidmadl, 1999
Hannover	9	80	27	Annual period	Herrmann and Hesse, 2002
Hannover	9	80	9	Possible	
ATV	150	--	36		ATV, 1995
SCST	**5.2**	**4891**	**25**	**LCA model**	

Pretreatment

The faeces and biowaste have to be pretreated to obtain a homogenous substrate for the digestion process. Faeces are passing an automated rake to eliminate interfering objects disposed in the toilet. Biowaste from kitchen and especially loppings have to be shredded in smaller parts, and potential hazardous substances (metals, plastics etc) are eliminated in a wet separation process (“swim-and-sink-process”) combined with a pulper. The energy demand of the different processing steps is summarized in Table 39. After the pretreatment and mixing of the substrates, the water content is adjusted to 90 - 97% if necessary.

Table 39: Energy demand for the pretreatment of digester substrate

Unit	Electrical energy demand	Source
Rake	0,07 kWh/m³	According to MURL, 1999
Shredder	1,2 kWh/Mg (1)	According to Vogt et al., 2002
Exhaust air treatment	8,1 kWh/Mg (2)	According to Vogt et al., 2002
Wet processing/ pulper	150 kWh/Mg (2)	According to Vogt et al., 2002

1) related to fresh matter of waste 2) related to dry matter

Hygienisation

The thermal hygienisation of the substrate is ensured by a pasteurisation prior to the digester. Pathogenic microorganisms are inactivated by means of elevated temperature (ca. 65°C) over a certain time period (one hour). The hygienisation step is usually applied ahead of the digester to minimize the risk of recontamination (ATV, 1996). The required thermal energy is produced by the CHP plant. Table 40 summarizes important parameters whilc calculating the thermal energy balance. A heat exchanger reclaims the thermal energy from the output flow of the hygienisation unit. If the thermal energy of biogas combustion is not sufficient for the hygienisation process, extra fuel (diesel) is added to the biogas plant.

Table 40: Parameters for the thermal energy balance of hygienisation

Parameters	Value	Unit	Source
Specific heat capacity of dry matter	1,05	MJ/(Mg*K)	MURL, 1999
Specific heat capacity of water	4,19	MJ/(Mg*K)	
Proportion of the recycled waste heat from the back-cooling of substrate to digester temperature	85	%	Assumption
Energy losses through transmission	5	%	According to sludge digestion (MURL, 1999)
Temperature of substrate after heat exchanger	35	°C	
Starting temperature of substrate (annual mean)	15	°C	Assumption
Hygienisation temperature	65	°C	

Digestion

The digester is operated in a mesophilic mode (40°C) with an average retention time of 30 days including subsequent stabilisation. The assumed operating conditions and the demand of energy and auxiliary materials are listed in Table 41. The expected biogas yield of the relevant substrates – faeces, urine (misdirected), biowaste and loppings – are stated in Table 42. The substrate is concentrated and stabilised in a closed secondary tank, where the biological processes are gradually stopped. The biogas which is still generated during this post-digestion process is also led to the CHP plant.

Table 41: Parameters of the mesophilic digestion process

Parameters	Value	Unit	Source
Retention time (Digestion and stabilisation)	30	days	
Water content digester	95	%	1)
Water content of dewatered residual sludge	65	%	1)
Water demand for solving of coagulation aid	200	kg/kg	1)
Dosage of coagulation aid	4	kg/Mg dry matter substrate	1)
Electrical energy demand of digester	3	kWh/Mg fresh mass	1)
Thermal energy demand of digester	0		2)
Electrical energy demand for dewatering	0,03	kWh/Mg dry matter residual sludge	1)

1) according to Vogt et al. 2002
2) energy transfer from hygienisation is sufficient to maintain operating temperature

Table 42: Biogas and methane yield of different substrates (related to input mass to digester)

Substrate	Organic dry matter (oDM) [% dry matter]	Biogas [m^3/kg oDM]	Methane [m^3/kg oDM]	Methane [kg/kg oDM]	
Faeces	93	0.45	0.29	0.21	1)
Urine*	75	0.34	0.22	0.16	2)
Biowaste	72	0.45	0.28	0.20	3)
Loppings	71	0.48	0.30	0.21	4)

* urine which was not separated in toilets
1) Estimation according to Otterwasser, 2005; assumed methane content in biogas 65%: Kujawa-Roeleveld et al., 2003 determined a methane content of 70% for blackwater digestion
2) Data from Otterwasser, 2005 suggest a significantly lower TOC proportion from organic dry matter in urine than in faeces; hence, the biogas yield is assumed to be smaller
3) Estimation according to mean values for kitchen biowaste in BLU, 2004
4) Estimation according to mean values for loppingskitchen biowaste in BLU, 2004

Prior to the composting, the stabilised digester residual is dewatered to 65% dry matter contents to establish aerobic conditions during the composting process. The separated wastewater contains some nutrients, salts and heavy metals (Table 43). The elution of heavy metals is estimated in analogy to the dewatering of sewage sludge. Other studies assume an average transfer of 50% of heavy metals into the wastewater (Vogt et al., 2002).

Open composting

The digester residual is further stabilized through open composting in piles. The relevant substance flows are based on the model in Vogt et al., 2002 (Table 44). The required time of composting to reach a sufficient degree of stabilization is set to 28 days. Piles are mechanically turned over biweekly, requiring the listed amount of diesel fuel.

Table 43: Transfer ratios in water during dewatering of residual

Substance	Transfer ratio [%]	Source
Phosphorus	15	(*)
Potassium	50	Vogt et al., 2002
Magnesium, Calcium	10	Vogt et al., 2002
Organic carbon	2	Vogt et al., 2002
Nitrogen, total	35.5	(*)
Ammonium-N	50	(*)
Heavy metals: estimated in analogy to sewage sludge dewatering		
Lead	20	
Cadmium	20	
Chrome	20	
Copper	15	
Nickel	40	
Mercury	20	
Zinc	25	

(*) elevated due to misdirected urine

Table 44: Substance flows of open composting of digester residual (open piles, 28 days retention time)

Input digester residual	Output	Source
Water content + water demand	Compost (stabilised): 40% water content	1)
	In off-gas: difference	
	Leachate: 22.5 L/Mg input	1)
Organic carbon	In compost: $C_{out} = 0.69 * C_{in}$	1)
	In off-gas: $CO_2\text{-}C_{out} = 0{,}290 * C_{in}$ $CH_4\text{-}C_{out} = 0{,}009 * C_{in}$ $NMVOC\text{-}C_{out} = 0{,}006 * C_{in}$	1)
Nitrogen	In compost: $N_{out} = 0.824 * N_{in}$	1)
	In off-gas: $NH_3\text{-}N_{out} = 0{,}170 * N_{in}$ $N_2O\text{-}N_{out} = 0{,}003 * N_{in}$ $N_2\text{-}N_{out} = 0{,}003 * N_{in}$	1)
Salts, Ca, P, etc, heavy metals and misc. inert substances	Assumption: salts and heavy metals accumulate in compost. Inert substances remain in compost	2), 1)
Diesel fuel (circulation)	2.2 L/Mg waste	1)

1) Vogt et al. 2002
2) Gronauer et al. 1997

Central heat and power plant

The generated biogas is combusted in a central heat and power plant (CHP plant) to gain electrical and thermal energy. A potentially required gas conditioning prior to the combustion process is neglected in this study. The amount of biogas lost by accidental leakage is set to 0.75% according to Ronchetti et al., 2002. Furthermore, the burning of excess gas should be allowed for in cases of system malfunction or storage overflow. Ronchetti et al., 2002 assume that 5% of the gas volume is flared in a modern biogas plant if complete usage of biogas for energetic purposes is intended, which is adopted in this study. Emissions from biogas flaring are rated the same as during combustion in the CHP plant.

Although pilot injection gas engines may have a higher efficiency and lower capital costs, the CHP plant in this study is equipped with a spark-ignition engine ("otto engine") in lean combustion mode with high excess air. Thus, legal emission limits can be reliably met. Table 45 summarizes CHP parameters and respective emission factors.

Table 45: Parameters and emission factors for CHP plant

		This study	Biogas-CHP	Natural gas-CHP
Source			Ronchetti et al., 2002	Calculated from database (IFU and IFEU, 2004)
Engine		60 kW Lean burn engine	60 kW Lean burn engine	50 kW (elec) Catalyst engine
Efficiency		32% electrical, 57% thermal	32% electrical, 57% thermal	29,3% electrical 58,6% thermal
Emission	**Unit**			
CH_4, combustion	mg/MJ	2,5	2,48	3,78
CO_2	mg/MJ	(*)	81.308	55.151
NO_x (as NO_2)	mg/MJ	38	37,85	62,98
N_2O	mg/MJ	1,6	--	1,57
CO	mg/MJ	51	50,93	51,17
SO_x (as SO_2)	mg/MJ	30	29,91	0,43
NMVOC	mg/MJ	2,5	2,48	4,72
Dust	mg/MJ	1,6	--	1,57

(*) depending on CO_2 and CH_4-input, less CO

3.3.2.4 Soil filter for greywater treatment

In two of the investigated alternative scenarios, greywater is treated in a planted soil filter. This natural treatment technology is based on the elimination of COD, nitrogen and phosphorus from wastewater during the passage of a soil filter that is planted with reed. The root system of the reed plants helps to prevent clogging of the soil filter, facilitates oxygen transport to the lower filter layers, and provides surface area for the growth of attached microorganisms. The contribution of the reeds to the removal of wastewater components is relatively small (Langergraber, 2005). Most of the COD and nitrogen is removed by microbial activity in the soil filter, while phosphorus is mainly retained by adsorption on soil particles (Wissing and Hofmann, 2002; Geller and Höner, 2003).

Process layout

Prior to the soil filter, particulate matter is removed by a sedimentation stage to reduce the pollutant load on the filter and prevent clogging by particle aggregation on the filter surface. The filter is operated as a vertical-flow soil filter with intermittent loading of greywater (Figure 15). Due to the insufficient long-term phosphorus elimination of the filter material, a post-precipitation stage is provided to remove the remaining phosphorus by precipitation with $FeCl_3$. Sludge from sedimentation and precipitation stage is dewatered and transported to an incineration plant. Due to the elevated heavy metal loads in greywater, sludge digestion or composting is not considered in this study to prevent the pollution of digester residual or compost with heavy metals. Reed plants are mowed once each year and added to biowaste treatment (composting or digestion). In case of composting scenario (Comp_Nat), faeces filtrate is added to particle-free greywater after sedimentation stage.

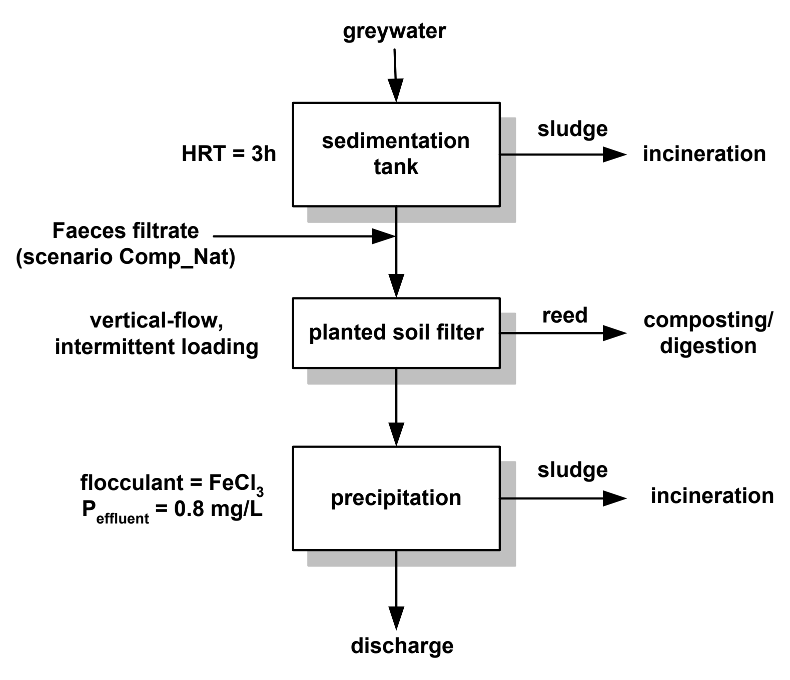


Figure 15: System layout for greywater treatment in soil filter

Process parameters

Drained greywater is lifted (pressure head = 4 m) and pumped to a sedimentation stage, where particulate matter of greywater (fraction of total TOC: 15%, N: 11%, P: 10%) is removed. The subsequent soil filter provides organic carbon removal, nitrification and partial denitrification, and phosphorus removal by adsorption (Table 46). P elimination can be as high as 90% in fresh filling material, especially if the iron content of the material is high (e.g. addition of iron sludge from drinking water treatment) (SCST, 2006). However, the long-term elimination rate for phosphorus is expected to reach 50% on average.

The loss of water through evapotranspiration from soil or plant surface can reach a significant fraction of the inflow. Depending on plants and weather conditions, evapotranspiration rates of 1-18 mm/d are possible. An average rate of 5 mm/d is assumed here. The energy demand of the soil filter is determined by the energy consumption of the feed pump. Beside the static pressure head (4m), a dynamic pressure head of 8m is considered according to the recommendations of ATV (ATV, 1998). Pumping of sedimentation and precipitation sludge is calculated via the water content of the sludge.

Table 46: Process parameters of planted soil filter for greywater treatment

Parameter	Unit		Source	Remarks
TOC	[% elimin.]	90*	Bahlo, 1999 Otterwasser, 2005	
NH_4-N	[% elimin.]	96*	Bahlo, 1999 Langergraber, 2005	
N total	[% elimin.]	40*	Otterwasser, 2005 Langergraber, 2005	partial denitrification
P total	[% elimin.]	50*	Bahlo, 1999 Otterwasser, 2005	adsorption, long-term average
Inorganic salts				elimination through plant uptake
Energy demand	[kWh/m³]	0.055	ATV, 1998	calc. with static and dynamic pressure head (4 + 8 = 12m)
Surface area	[m²/pe]	2	SCST, 2006 Oldenburg, 2002	1-3 m²/pe
Evapotranspiration	[L/(m²*d)]	5	Oldenburg, 2002 Langergraber, 2005	1-18 mm/d

(*) dissolved fractions, rate does not include elimination in sedimentation stage

Allocation of substance flows

The plant uptake of organic carbon, nutrients, and inorganic salts is estimated from literature (Table 47). By recycling of the mowed reed to the composting or digestion process, a small amount of nutrients and organic carbon can be reused as fertilizer. Organic matter of plants is formed from CO_2. The soil filter produces an average amount of 2.44 kg reed per m² and year above ground with a dry matter content of 41% (Peverly et al., 1995). Sludge production from microbial growth or clogging inside the soil filter is neglected.

Table 47: Plant uptake of greywater components in soil filter

Substance	Unit		Source	Remarks
Dry matter	[g/(m²*a)]	1000	Peverly et al., 1995	Plant production
C_{org}	[g/(m²*a)]	370		formation from CO_2
N	[g/(m²*a)]	10	Langergraber, 2005	small due to low influent N concentration
P	[g/(m²*a)]	3	Langergraber, 2005	
K	[g/(m²*a)]	10		estimated
S, Ca	[%]	1		estimated
Cl, Na	[%]	0		estimated

The distribution of nitrogen species in the effluent of the soil filter is determined according to conventional wastewater treatment due to lack of suitable data. Due to the low nitrogen content in greywater, this assumption has a small influence on the overall evaluation.

The elimination of heavy metals in the sedimentation stage is neglected. Their removal in the soil filter occurs mainly by adsorption on soil particles. The accumulation of heavy metals in plants is estimated to be negligible (Table 48). Most of the heavy metals seem to accumulate in the roots, which are not harvested and remain in the soil filter (Vymazal and Krasa, 2003). Elimination efficiencies have been investigated for Cu and Zn. For other heavy metals, elimination has been roughly estimated to be equivalent to conventional wastewater treatment. The disposal of loaded soil filter material after a certain operating period is not considered within this study.

Table 48: Heavy metal transfer coefficients in soil filter

SOIL FILTER					POSTTREATMENT
Substance	Effluent	Filter material	Plant uptake (*)	Source	Additional elimination (**)
	[%]	[%]	[%]		[% of filter effluent load]
Pb	20	79	1	(1)	8
Cd	30	69	1	(1)	7
Cr	20	79	1	(1)	8
Cu	20	79	1	(2, 3)	8.5
Ni	40	59	1	(1)	6
Hg	20	79	1	(1)	8
Zn	20	79	1	(2,3)	7.5

(*) above-ground, roughly estimated from Geller and Thum, 1999
(**) estimated elimination is 10% of activated sludge process (see annex)
(1) estimated (Fuchs et al., 2002)
(2) Vymazal and Krasa, 2003
(3) Otterwasser, 2005

In the post-precipitation stage, ferric chloride ($FeCl_3$) is dosed as a flocculant to eliminate the remaining phosphorus content of the soil filter effluent to a concentration of P = 0.8 mg/L (equivalent to chemical P elimination in conventional wastewater treatment, $\beta = 1.5$). Other components of soil filter effluent are supposed to be partially eliminated as well: it is estimated that 5% of remaining carbon, organic nitrogen and sulphur are transferred into the precipitation sludge. Likewise, the fraction of heavy metals bound in the sludge is estimated to 10% of the elimination in activated sludge process (see Table 48). These rough estimations may differ considerably from the real transfer coefficients in a post-precipitation stage. However, the impact of these assumptions is negligible for the evaluation despite the P elimination, for which the post-treatment is designed.

Table 49 summarizes the overall transfer coefficients for organic carbon and nutrients for the complete greywater treatment, including sedimentation, planted soil filter, and post-precipitation. The implementation of a post-treatment step for P elimination implies chemical demand and increased sludge production. The impacts of greywater treatment with and without post-treatment will be investigated in sensitivity analysis.

Table 49: Transfer coefficients in greywater treatment with soil filter (*)

Input	Emission as	Effluent	Air	Reed	Filter material	Sludge
		[%]	[%]	[%]	[%]	[%]
Organic carbon	TOC	8.1				
	CO_2-C		76.5			
	TOC in sludge					15.4
N_{total}-N	NH4-N	1.8				
	NO3-N	31.0				
	organic N	19.6				
	NH_3-N		0.3			
	N_2O-N		0.1			
	N_2-N		31.0			
	N in reed			4.2		
	N in sludge					12.0
P_{total}-P	PO4-P	11.2				
	P in reed			3.3		
	P in filter material				41.7	
	P in sludge					43.8
K	K	97.3				
	K in reed			2.7		

(*) including sedimentation, soil filter, and post-precipitation, scenario VAC_NAT

3.3.2.5 Sequencing batch reactor for greywater treatment

As an alternative to natural greywater treatment in a planted soil filter, two of the alternative scenarios consider the treatment of greywater in a technical process. This process is laid out as a sequencing batch reactor (SBR), an activated sludge process with nitrification, partial denitrification and chemical P elimination. The different phases of the process (filling, aeration and mixing, sedimentation, discharge) proceed in a single reactor within a temporal sequence. At the end of the cycle, excess sludge is withdrawn and aerobically stabilized through continuous aeration (sludge age: 25 d). Stabilized sludge is thickened with flocculation aids and dewatered in a centrifuge to a dry matter content of 40%, before it is incinerated in a waste incineration plant (truck transport: 30 km). The process layout is depicted in Figure 16. The process model is adapted from conventional wastewater treatment as described in chapter 3.3.1.2. However, the different composition of greywater requires some adjustments of the model. Whereas the distribution of nitrogen and phosphorus to the different output flows is calculated similar to conventional WWTP, excess sludge production is estimated via a simplifying factor.

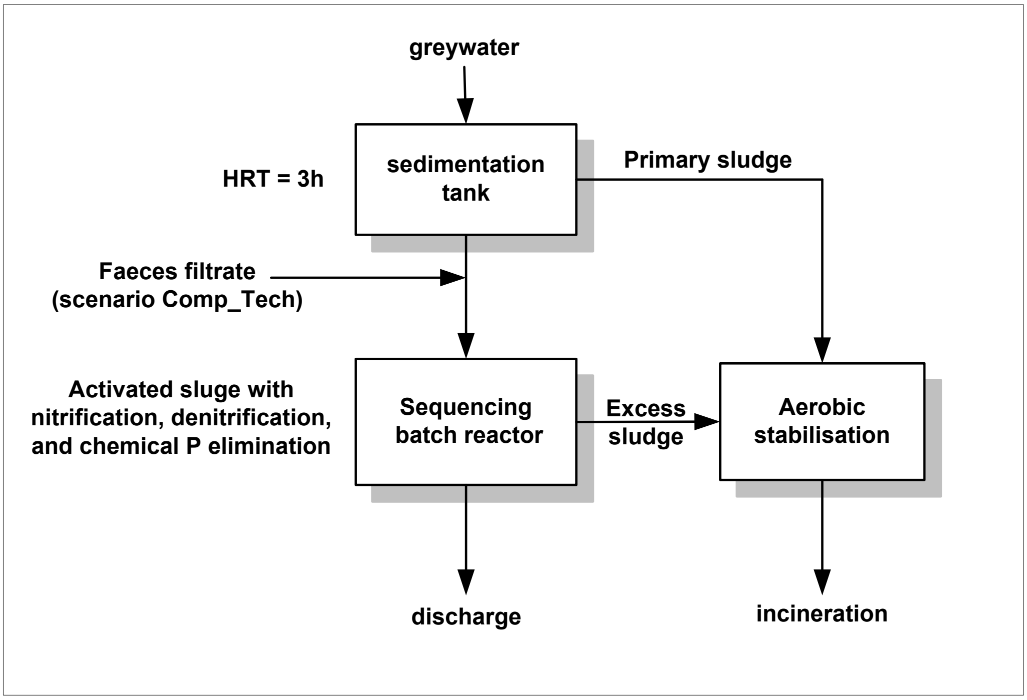


Figure 16: System layout for greywater treatment in sequencing batch reactor

Table 50 lists the basic process parameters of the SBR process for greywater treatment. Particulate matter is separated in the sedimentation tank (fraction of total TOC: 15%, N: 11%, P: 10%), and faeces filtrate is added after the primary treatment if required. Nitrogen is partially eliminated by nitrification and denitrification (mixing without aeration), while phosphorus is eliminated simultaneously by addition of a flocculant ($FeCl_3$), resulting in an effluent concentration of 0.8 mg P/L. Biological phosphorus elimination accounts for 20% of the total P removal.

Table 50: Process parameters of SBR for greywater treatment

Parameter	Unit		Remarks
TOC	[% elimination]	93*	assumed process efficiency for carbon and nitrogen elimination (see Helmreich et al., 2000; Steinmetz et al., 2002)
NH_4-N	[% elimination]	96*	
N total	[% elimination]	70*	
P total	[% elimination]	86*	20% Bio-P, $P_{effluent}$: 0.8 mg/L
Heavy metals	[% elimination]	60-85	according to conventional WWTP
Inorganic salts	[% elimination]	0	neglected
Sludge age	[d]	25	according to ATV, 1997
Sludge production	[g dry matter/g TOC]	0.6	70% of dry matter are organic matter with 50% carbon content
Energy demand	[kWh/m^3]	0.44	scenario Vac_Tech, calculated according to conventional WWTP
	[kWh/m^3]	0.5	scenario Comp_Tech, elevated due to faeces filtrate

(*) dissolved fractions, rate does not include elimination in sedimentation stage

Heavy metal elimination is estimated to be similar to conventional wastewater treatment, and the incorporation of inorganic salts into the sludge is neglected here. Transformation of sludge components and possible emissions from aerobic stabilisation are not considered either. The recycling of water from sludge dewatering to the influent of the SBR process has to be neglected due to the linear and non-recursive model structure. However, volume and nutrient loads of sludge water are < 2% of the total influent volume and <1% of the influent nutrient load, so the simplification has a small influence on the process. In the model, sludge water is discharged with effluent.

The energy demand is calculated from the particular processes (greywater lifting, aeration, mixing, precipitation, sludge dewatering, and auxiliaries) according to conventional WWTP (see annex). The transfer coefficients for the greywater treatment in an SBR process are listed in Table 51, including the processes of sedimentation, sequencing batch reactor, sludge stabilisation and dewatering.

Table 51: Transfer coefficients of elemental flows in greywater treatment with sequencing batch reactor*

Input	Emission as	Effluent [%]	Air [%]	Sludge [%]
TOC	TOC	6.0		
	CO_2-C		62.4	
	C_{org} in sludge			31.6
P_{total}-P	P-species	12.6		
	P in sludge			87.4
N_{total}-N	NH_4-N	1.8		
	NO_3-N	15.0		
	N-species	10.0		
	N_2-N		39.7	
	NH_3-N		0.3	
	N_2O-N		0.1	
	N in sludge			33.1

(*) including sedimentation, SBR, and sludge dewatering; scenario Vac_Tech

3.3.3 System expansions

Two processes are considered for expanding the investigated scenarios to allow for proper system comparison: the supply of energy (electrical and thermal) and the industrial production of mineral fertilizer. The basics of system expansion are explained in chapter 2.1.4. Here, the respective processes are described in detail. To calculate the amount of energy and fertilizer that has to be additionally supplied in a certain scenario, a balance is set up where the respective energy or fertilizer outputs of all scenarios are compared. All scenarios are related to the scenario which supplies the most fertilizing equivalents or energy, and the difference has to be provided by system expansion processes.

3.3.3.1 Energy supply

The supply of additional electrical or thermal energy for system expansion purposes is described with the basic models for energy supply (see chapter 3.1.1). Electrical energy can be transported without major losses, which makes it suitable for an effective substitution. Thermal energy exhibits significant losses depending on transport conditions and application. Hence, surplus thermal energy from a certain process might not be transferred to other possible consumers in reality, if conditions are not profitable. However, thermal energy plays only a minor role in the overall energy balance of this LCA study, and it is assumed that surplus thermal energy can be utilized at the point of emergence.

3.3.3.2 Industrial fertilizer production

Numerous different products of single or multi-nutrient fertilizers are industrially produced and applied by farmers, and they offer a range of different nutrient contents. The main nutrients are nitrogen (as N), phosphorus (as P_2O_5), potassium (as K_2O), calcium (as CaO) and magnesium (as MgO). Beside the valuable nutrient content, industrial fertilizers contain considerable amounts of heavy metals, originating from raw materials (e.g. raw phosphate ores) and possibly enriched during the production process. For this study, it is important to characterize the substitutable mineral fertilizers in terms of resource usage and emissions in their production process and regarding their heavy metal content.

For calculating an average heavy metal content of a certain nutrient fertilizer, the nutrient concentration and market share of the different fertilizers have to be connected with their average heavy metal content. Patyk and Reinhardt give an overview of the nutrient content of different fertilizers (Patyk and Reinhardt, 1997). The respective market shares in the year 1998/99 are taken from Drescher et al. for the main fertilizer types (Drescher-Hartung et al., 2001). The average heavy metal contents are compiled from several studies (Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001) which are essentially based on the work of Boysen (Boysen, 1992). Table 52 shows the calculated heavy metal contents of industrial fertilizers related to the respective nutrient content. Detailed documentation for this calculation is given in the annex.

Table 52: Mean concentrations of heavy metals and As for average mineral fertilizers, related to the single nutrients

Values in mg/kg nutrient	As	Cd	Cr	Cu	Ni	Hg	Pb	U	Zn
N-fertilizer (as N)	9,3	6,0	77,9	26,0	20,9	0,07	54,9	51,5	203,0
P-fertilizer (as P_2O_5)	14,5	39,5	543,2	90,5	88,3	0,3	67,0	349,2	839,2
K-fertilizer (as K_2O)	0,1	0,1	5,8	4,8	2,5	0,03	0,8	1,0	6,2

The contamination of phosphate fertilizer with elevated levels of Cd, Cr, and Zn is well-known and originates from the raw phosphate ores. Recently, the toxic element uranium (U) was detected in significant concentrations in mineral phosphate fertilizers (Kratz, 2004; Fink, 2005). Uranium is not included within the impact assessment method of this study, but due to its toxicity and potential accumulation in the soil it is addressed separately in the results.

The production of mineral fertilizers and the associated substance and energy flows are documented in detail by Patyk and Reinhardt (Patyk and Reinhardt, 1997). However, emissions in surface waters are not considered within their study, although they can play an important role for the environmental evaluation (e.g. phosphate and fluoride emissions in processing of raw phosphates). Hence, aquatic emissions are adopted from a Suisse study (Gaillard et al., 1997) and recalculated, relating them to the average single-nutrient fertilizer via market shares and nutrient content. Table 53 shows an abstract of important LCI data regarding the production process of mineral fertilizers. The data takes into account the complete processes of production and supply of mineral fertilizers, including transport and energy supply, starting with the extraction of resources until the packing of the marketable product.

The production of nitrogen fertilizer requires large amounts of fossil energy for the fixation of nitrogen (“Haber-Bosch-process”) and emits related amounts of flue gases (N_2O, NO_x, CO_2). The P-fertilizer production causes airborne and aquatic emissions of fluoride and phosphate from the multi-step processing of phosphate ore and from the dumping of production wastes (phosphogypsum stacks). Details of data collection are provided in the annex.

Table 53: Life cycle inventories of mineral fertilizer production (abstract) from Patyk and Reinhardt, 1997 and Gaillard et al., 1997

Reference value		**1000 kg N**	**1000 kg P_2O_5**	**1000 kg K_2O**
Input	Unit			
Use of ressources				
Raw potash	kg			10,500
Limestone	kg	550		
Raw phosphate ore	kg		4,060	
Sulphur	kg		272	
Cum. energy demand (fossil)	MJ	48,264	16,337	9,866
Output				
Emissions (air)				
Dust (>PM10)	kg	2,31	1,11	0,85
NH3	kg	6,69	0,01	0,00
N2O	kg	15,05	0,04	0,05
HF	kg	0	0,023	0
CO2, fossil	kg	2.820	1.117	617
CO	kg	2,80	1,42	0,42
NOx	kg	15,76	8,58	1,15
SO2	kg	5,16	11,98	0,27
CH4	kg	7,45	2,07	1,38
NMVOC	kg	0,54	0,49	0,12
Emissionen (water)				
Metals				
Al	g	476,09	94,71	23,4
As	g	0,96	4,59	0,05
Cd	g	0,03	4,40	0,00
Cr	g	4,94	23,04	0,28
Cu	g	2,40	22,47	0,12
Ni	g	2,43	18,11	0,12
Hg	g	0,00	4,18	0,00
Pb	g	2,67	19,58	0,19
Zink	g	4,95	27,48	0,27
Nutrients				
NH3	g	2,68	9,17	1,72
NO3	g	189,15	8,16	1,20
PO4	g	28,62	4400	1,40
Chlorid	g	6219,00	5826,89	825,00
Fluorid	g	1,65	2200	0,27

3.3.4 Fertilizer application

During the application of different types of fertilizers, emissions arise from both the fertilizers themselves and the operation of the agricultural tractor. Additionally, the tractor requires diesel fuel for operation. However, most of the fertilizing equivalents supplied by the alternative systems in this study derive from the application of urine as a secondary fertilizer. Its equivalent in the reference scenario is the industrially produced mineral fertilizer. Compost (from biowaste, faeces, or digester residual) plays only a minor role in the systems´ overall nutrient input into agriculture. Besides, the total mass flow of compost represents only a minor fraction from that of diluted urine (< 5%). It often serves as a soil conditioner and is not explicitly applied for short-term nutrient supply. Hence, the energetic expenditures and machinery emissions from compost application are neglected in this study. Fuel demand and emissions of the agricultural tractor is only determined for the application of urine and mineral fertilizer. Airborne emissions from the evaporation of fertilizer components are considered for each fertilizer in this study (Figure 17).

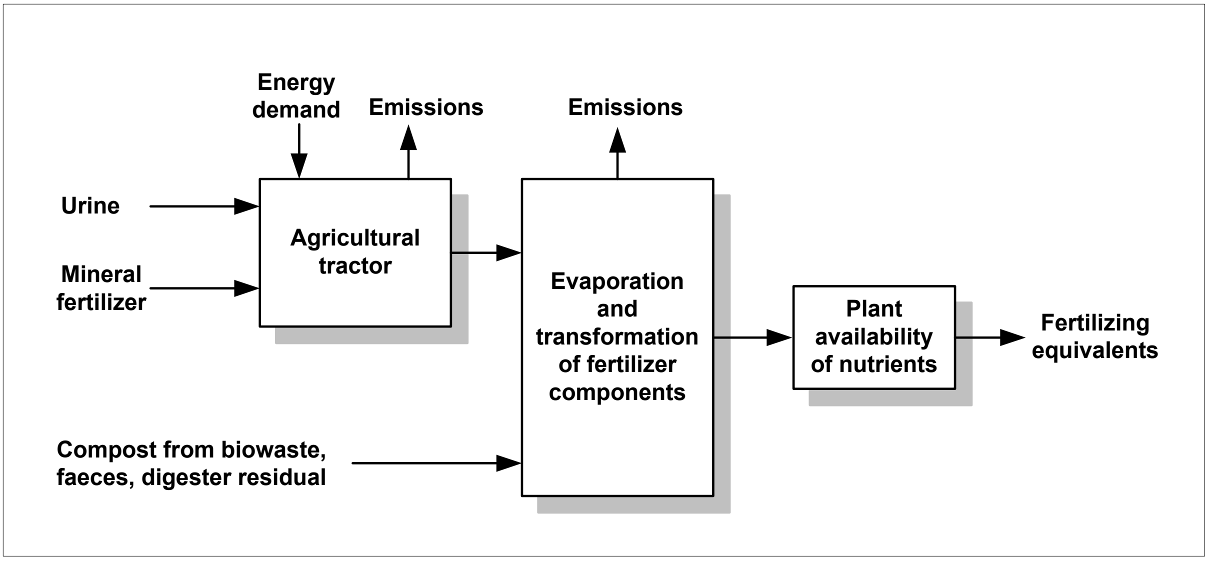


Figure 17: Processes during fertilizer application

For the calculation of the respective emissions and energy demand, the first step is to determine the acreage on which the urine is applied. Through the working time of the tractor in different loading levels of the engine, the fuel demand and the airborne emissions of the agricultural machinery can be calculated accordingly. Finally, the type of fertilizer and the application technology determine the airborne emissions of fertilizer components during application. The effective fertilizing equivalents of the respective fertilizer are then estimated with the average plant-availability of the nutrients.

Calculation of the acreage for fertilizing

From exemplary data for winter wheat, the following fertilizer doses are necessary for its cultivation (Finck, 1992):

P_2O_5:	90 kg/ha, one dose per annum
K_2O:	160 kg/ha, one dose per annum
N:	200 kg/ha, split into four doses per annum

Table 54 lists a suggestion for the distribution of the required nutrients on different types of fertilizer, taking into account the approximate nutrient content.

Table 54: Distribution of the nutrient amounts of manure, mineral fertilizer and urine for the cultivation of winter wheat

Nutrient	**Demand** [kg/ha*a]	**Manure** [kg/ha*a]	**Mineral fertilizer** [kg/ha*a]	**Urine** [kg/ha*a]
P_2O_5	90	67	0	27
K_2O	160	95	34	31
N	200	90	0	110

The urine separation process results in a total volume of 2008 m^3 per year and contains 12323 kg nitrogen, 1339 kg phosphorus, and 3481 kg potassium as plant-available nutrients. Assuming the fertilizer management from Table 54, the separated urine can be applied to an acreage of ca. 112 ha. The volume of applied urine amounts to around 18 m^3/ha, which is applied in two doses annually. Together with dilution water, the volume of this liquid fertilizer is around 30 m^3/ha .

Working time of agricultural tractor

Solid mineral fertilizers are applied with various methods (e.g. centrifugal spreader). Liquid fertilizers are applied with splash plates, nozzles, drag hoses, or liquid injection. To minimize nitrogen losses during application, drag hoses or injection systems should be applied for urine or the liquid fertilizer should be immediately incorporated.

Table 55 lists the estimated working time and the distribution of engine loading levels for the application of urine or mineral fertilizer. Data is compiled from charts in Rinaldi et al., 2005. The respective values are set in the module “agricultural tractor” contained within the software UMBERTO® (IFU and IFEU, 2004). This module calculates the energy demand and the emissions from agricultural tractor operation depending on engine power, working time, and engine loading levels, and is based on data from Borken et al., 1999.

Assuming that mineral fertilizer needs three applications a year (two for nitrogen and one for phosphorus and potassium), the respective working time amounts to

$$3 * 0.7 \text{ h/ha} * 112 \text{ ha} = 235 \text{ h}$$

For the urine application, nutrients are applied as a combined dose with two applications per year, resulting in a total working time of

$$2 * 1.6 \text{ h/ha} * 112 \text{ ha} = 358 \text{ h}$$

It has to be noted that due to the large volume the fertilizing with urine needs a 50% longer working time for the farmer. However, the considerations of this rough calculation have to be checked in reality.

Table 55: Working time and distribution of engine loading levels during the application of mineral fertilizer and urine (according to Rinaldi et al., 2005)

Type of fertilizer		Mineral fertilizer	Urine
Nominal power of tractor	[kw]	50	50
Device		Spreader width: 15 m	Pressure drum: 6.5 m³
Working time (*)	[h/ha]	0.7	1.6
Applied volume	[m³/ha]	--	30
Distribution			
Heavy loading	[%]	10	10
Medium loading	[%]	60	50
Road	[%]	20	30
Engine idle	[%]	10	10

(*) including all working steps, road traffic, etc.; field size: 2 ha, distance farm to field: 1000 m

<u>Emissions during application</u>

Mineral fertilizers are delivered as solid stable chemicals without the potential for gaseous emissions. However, their application on agricultural fields may cause emissions of NH_3, N_2O, NO_x, and CO_2 through hydrolysis and various chemical reactions. The rate of emissions is influenced by the chemical composition of the fertilizer and soil parameters (pH, moisture, composition) (ECETOC, 1994). A simplified calculation of emissions can be performed with average emission factors (EMEP/CORINAIR, 2004; see Table 56). Emission factors for certain fertilizers are connected to nutrient content and market share to calculate these average factors. The carbon dioxide emitted from urea hydrolysis originates from fossil sources and contributes to climate change.

The application of secondary fertilizers like urine or compost is connected with emissions as well. NH_3 emissions of urine application are estimated from various pilot studies, while N_2O and NO_x emission factors are estimated to be equivalent to those of mineral fertilizer (Table 56). For compost, assumptions from an LCA study of biowaste composting are adopted. Carbon dioxide from urea hydrolysis of urine and from carbon degradation of compost does not contribute to climate change due to the regenerative source of the carbon (= human food).

Table 56: Emission factors for mineral and secondary fertilizers during agricultural application

	NH_3 g NH_3-N/g N	N_2O g N_2O-N/g N	NOx g NO-N/g N	CO_2 fossil g CO_2/g N	CO_2 regen. g CO_2-C/g C	Source
Mineral N fertilizer	0.05	0.0125	0.007	0.59	0	(1,2)
Urine	0.06	0.0125	0.007	0*	0	(3,4,5)
Compost from biowaste/faeces	0.05	0.0125	0.007	0	1**	(6)
Compost from digester residual	0.063	0.0125	0.007	0	1**	(6)

(*) CO_2 from urea hydrolysis is from regenerative sources (human food)
(**) long-term degradation of 50% of organic carbon content
(1) ECETOC, 1994
(2) EMEP/CORINAIR, 2004
(3) Stockholm Vatten, 2000
(4) Vinneras et al., 1998; Vogt et al., 2002
(5) SCST, 2006
(6) according to biowaste composting in Vogt et al., 2002 (NH_3 emissions: 37% of NH_4-N and 4% of Non-NH_4-N, NH_4-N-content of biowaste compost: 3% of total N, NH_4-N-content of composted digester residual: 7% of total N, rest is Non-NH_4-N))

Plant availability

The plant availability of the nutrients (N,P,K) is a decisive factor to determine the substitution potential of secondary fertilizers like urine and compost. Mineral fertilizers normally supply their total amount of contained nutrients in a short term for plant growth. The short-term availability is of particular importance with nitrogen. Secondary fertilizers may cover the short-term nutritional requirement only partially, because they provide nitrogen at a slower rate. For phosphorus and potassium, the short-term availability is of minor importance. However, fractions of P and K content may not be plant-available even in the medium or long term due to their chemical fixation. Factors influencing the nutrient availability are e.g. utilisation, nutritional effect, availability, C/N ratio, C/P ratio, etc. (see annex for details).

In this LCA study, it is assumed that 100% of nutrients in mineral fertilizers are plant-available in a short term. The relative availability of nutrients in secondary fertilizers is defined according to pilot studies (Table 57). However, more research is needed to exactly determine the nutritional substitution potential of secondary fertilizers.

For urine, the availability of all nutrients seems to be equal to that of mineral fertilizer. Results from pot and field tests showed no significant difference to industrial fertilizer in terms of nitrogen availability (SCST, 2006). For compost from biowaste, faeces, or digester residual, factors from other studies are adopted. The limited availability of nitrogen and phosphorus from compost has been observed in several studies and is originating from the considerable fraction of organically bound N and P.

Table 57: Plant availability of nutrients from secondary fertilizers with regard to the substitution potential for mineral fertilizer

	Nitrogen [% of total N]	Phosphorus [% of total P]	Potassium [% of total K]	Source
Urine	90-100 / 60-90 / 80-90 **100**	-- / 80-100 / -- **100**	-- / -- / -- **100**	1 / 2 / 3
Compost	50 / 10 / 5-15 **10**	100 / 100 / 20-40 **70**	100 / 100 / >85 **100**	4 / 5 / 6

Bold : this study
(1) SCST, 2006
(2) Bengtsson et al., 1997
(3) Stockholm Vatten, 2000
(4) assumptions in Vogt et al., 2002
(5) EPEA, 2004
(6) Stadtmüller, 2004

3.3.5 Urine treatment

In the following, different processes for the treatment of urine are described. Targets are volume reduction, improved handling, and inactivation of micropollutants. Even though some research has been conducted to investigate possible processes (Maurer et al., 2006), the process data for energy and material demand and process efficiencies has not been stated for an industrial scale. For a first assessment of the process alternatives, data from laboratory or pilot-scale experiments is used together with reasonable assumptions.

For the transport of urine from collection tanks to the treatment facility by tanker truck, a distance of 10 km is estimated. Fur the subsequent transport of the fertilizer products from urine treatment to farms, the respective distance is assumed to be another 10 km (equal to the distance from collection tank to the storage facility at the farms).

Struvite precipitation and ion exchange

The process of struvite precipitation and subsequent ion exchange with zeolithes has been investigated in laboratory experiments in Sweden (Ganrot, 2005). For the precipitation of struvite, magnesium oxide (MgO) is added to the urine in a small stoichiometric excess (0.7 kg/m^3 urine). The kinetics of the precipitation process is very fast and takes only a few seconds. The increased pH of stored urine (~ pH 9) further supports the precipitation process, and the phosphorus content of urine can be precipitated quantitatively. To recover the remaining nitrogen content (mainly NH_4-N), zeolithes are added to the urine (20 kg/m^3), which act as strong adsorbents for NH_4-N. After a longer reaction time (several hours), the precipitated struvite and the loaded zeolithes are removed by a centrifugal process, before they are dried and used as fertilizer. The energy demand of the separation process is adopted from Maurer (5.6 MJ/kg N in struvite, equivalent to 0.23 MJ/kg separated solids (Maurer et al., 2003)). The remaining depleted urine is ozonated for inactivation of micropollutants, before it is treated in a biological process (SBR) to remove remaining nitrogen and COD. The process model for the SBR plant is adopted from the conventional scenario, including sludge disposal in an incineration plant. The important parameters of the process combination are depicted in Figure 18.

The overall efficiency for the recovery of nutrients is 99% for phosphorus, 90% for nitrogen, and 30% for potassium. According to the researchers, the product is a slow-release fertilizer with good plant availability. However, the process has not been investigated yet on a pilot-plant or even industrial scale.

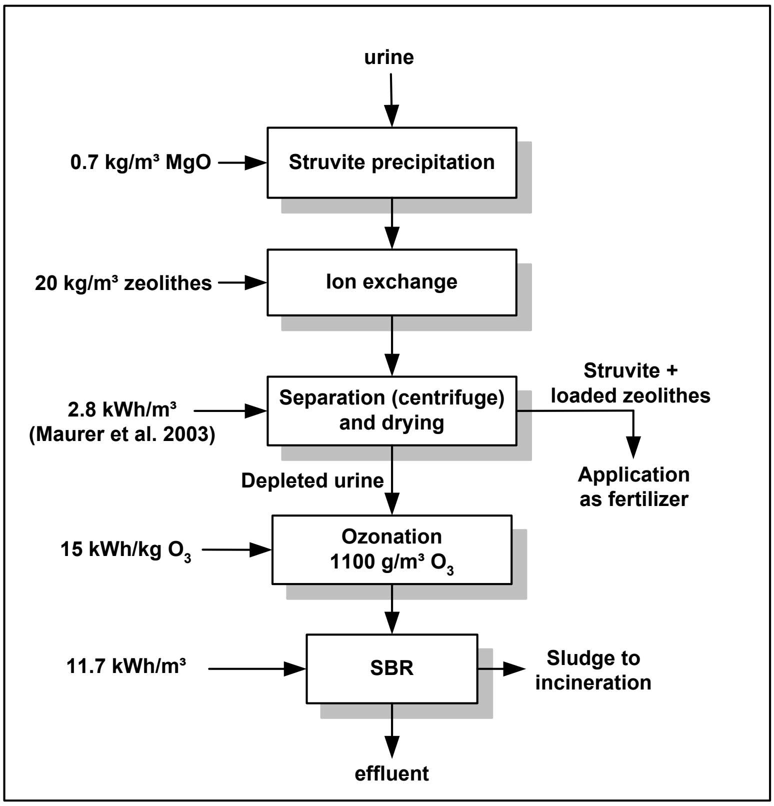


Figure 18: Process layout of urine treatment by struvite precipitation and ion exchange

Steam stripping, ozone, evaporation

Another process for the recovery of nutrients from separated urine has been investigated in a part of the SCST project (SCST, 2006). It is based on the physical removal of NH_4-N as ammonia gas by stripping the urine with steam. The condensate contains 90% of the urine-N and can be utilized directly as a fertilizer. The depleted urine is treated with ozone to inactivate micropollutants, before its volume is further reduced by evaporation. A tenfold concentration of the remaining nutrients seems to be achievable. The product is a viscous liquid with high concentrations of nitrogen, phosphorus, potassium, and other trace elements. Crystallisation of MAP or related salts can occur spontaneously. The distillate of the evaporation process is supposed to contain small amounts of NH_3 (5% of substrate-N) and volatile organic acids (10% of substrate-C) and has to be treated in a biological process (SBR).

The described processes could only be investigated separately from each other, so that the combination of processes is not yet examined. However, process data for a possible treatment process combination has been estimated in close collaboration with the project partners. Figure 19 shows the process layout and important parameters. The energy demand is crucial for the evaluation of this process combination, as both steam stripping and evaporation require large amounts of electrical and thermal energy. Combining the processes, it was postulated that 30% of the total energy demand could possibly be saved (e.g. the urine is preheated from steam stripping and thus can be evaporated with less energy demand).

The overall efficiency for the recovery of nutrients is 100% for phosphorus, 96% for nitrogen, and 100% for potassium.

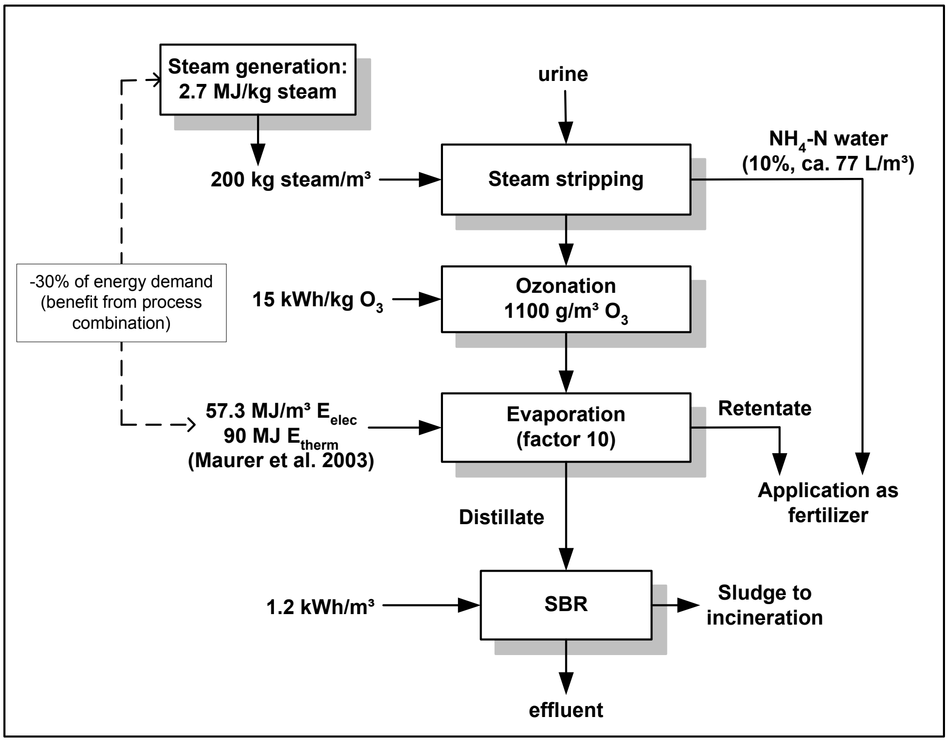


Figure 19: Process layout of urine treatment by steam stripping and evaporation

Struvite precipitation and steam stripping

The third process combines struvite precipitation and steam stripping. The addition of MgO to urine leads to the precipitation of struvite, and phosphorus can be recovered quantitatively. The remaining nitrogen is recovered by subsequent stripping with steam. The high pH value after struvite precipitation (mainly from MgO addition) supports the conversion of NH_4 into ammonia and thus increases the efficiency of the process. Struvite is separated from the depleted urine by a centrifuge and dried (energy demand: 0.23 MJ/kg dried solids (Maurer et al., 2003)). Dried struvite and the condensed ammonia water can be used as fertilizers. The depleted urine is ozonated and treated in a biological process (SBR) to remove remaining nitrogen and COD. The process layout and relevant data is presented in Figure 20.

The overall efficiency for the recovery of nutrients is 90% for phosphorus, 95% for nitrogen, and 30% for potassium in this process combination.

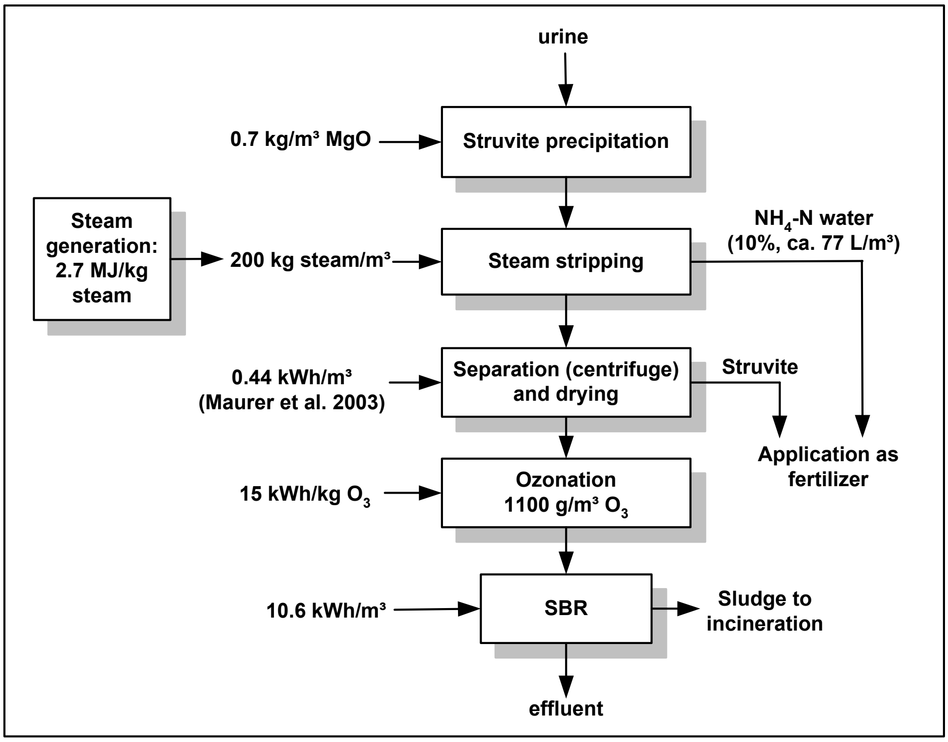


Figure 20: Process layout of urine treatment by struvite precipitation and steam stripping

4 Results and discussion

This section of the report describes the results of the LCA study. At first, system expansion processes (supply of industrial fertilizer and energy) are calculated for each scenario. Selected results of the Life Cycle Inventory (LCI) are presented to follow the flows of nutrients and heavy metals in the different sanitation systems. These LCI results form a basis for the subsequent evaluation of ecological indicators via Life Cycle Impact Assessment (LCIA). The different indicators are weighted by normalisation to the respective indicator values in inhabitant equivalents (reference: D 2003). Thus, all indicators can be summarized to form an overall ecological profile ("eco-profile") to allow conclusions for the ecological comparison. Selected results of the sensitivity analysis show the influence of certain system parameters on related indicators. Relevant key parameters for the LCA study of sanitation systems can thus be identified. Finally, the results of the separate LCA assessment of urine treatment options complete this section.

4.1 System expansion

4.1.1 Fertilizer

Table 58 lists the amounts of fertilizer that each scenario supplies for agriculture. The relevant nutrients N, P, and K are calculated from the nutrient content of the different substance flows, the losses during processing, and from plant-availability of the nutrients. Each scenario is expanded by the listed amount of industrially produced fertilizer to equalize the supplied fertilizing equivalents. In the reference scenario, some nutrients are provided from the regular composting and agricultural application of biowaste. However, most of the nutrients have to be produced industrially (99% of N, 78% of P, 68% of K). The alternative scenarios supply these nutrients via secondary fertilizers. Urine separation and application generates 96% of total N, 50% of total P, and 48% of total K, emphasizing the large fertilizing potential of urine. Composted faeces are only relevant for the phosphorus (45-49%) and potassium supply (31-51%), whereas the plant-available nitrogen content of compost is negligible. Beside the considerable losses of nitrogen via airborne emissions during the composting process, most of the nitrogen is organically bound and cannot be readily taken up by plants.

In relation to the inhabitants of the settlement, the alternative sanitation scenarios can generate annual fertilizing equivalents of up to 2.6 kg N, 0.54 kg P, and 1.5 kg K per person and year. Urine plays a major role in the substitution of mineral fertilizers, whereas composted faeces amend the supply of phosphorus and potassium. Again, it has to be mentioned that the benefits of organic carbon for soil fertility are not accounted for in this LCA study.

Table 58: Fertilizer supply from compost, urine, and industrial fertilizer (only plant-available nutrients)

Scenario	Ref		Comp_Nat		Comp_Tech		Vac_Nat		Vac_Tech	
Nutrient	[kg/a]	[%]	[kg/a]	[%]	[kg/a]	[%]	[kg/a]	[%]	[kg/a]	[%]
N compost	**140	1	322	3	312	3	490	4	481	4
N urine	0	0	12323	96	12323	96	12323	96	12323	96
N industrial*	12673	99	168	1	178	1	0	0	9	0
N total	12813	100	12813	100	12813	100	12813	100	12813	100
P compost	**584	22	1182	45	1162	44	1311	49	1294	49
P urine	0	0	1339	51	1339	50	1339	51	1339	51
P industrial*	2066	78	129	4	149	6	0	0	17	0
P total	2650	100	2650	100	2650	100	2650	100	2650	100
K compost	**2350	32	3840	52	3743	51	2330	32	2281	31
K urine	0	0	3481	48	3481	48	3481	48	3481	48
K industrial*	4971	68	0	0	97	1	1510	20	1559	21
K total	7321	100	7321	100	7321	100	7321	100	7321	100

* system expansion (see 2.1.4)
** from biowaste compost

4.1.2 Energy

The supply of electrical and thermal energy can be another secondary function of the alternative sanitation systems. An input-output balance for electrical and thermal energy is set up to show the allocation of the energy demand and determine the surplus amount of energy that is generated by SCST systems.

Electrical energy

Table 59 lists the electrical energy demand for the various processes of the sanitation scenarios and their share of the total electrical energy demand. The difference between input and output describes the effective electrical energy demand of the scenarios, which differs significantly between the scenarios. The system expansion process equalizes the different outputs of electrical energy of the scenarios.

In the reference scenario, the energy demand is mainly determined by drinking water supply and wastewater treatment. A fraction of this energy demand can be balanced by the energy benefits of sewage gas and biowaste incineration. Concerning the SCST systems, the composting scenarios need less energy input than the digestion scenarios (Figure 21). However, the energy benefits from biogas generation and combustion are considerable and offset the high energy demand of the vacuum system. Natural greywater treatment in a soil filter can further decrease the required energy for water treatment by a factor of 8-9. Consequently, the alternative scenario with the smallest effective energy demand is scenario Vac_Nat. The energy savings from the reduction of drinking water demand by the vacuum system further contribute to this trend, although they do not play a major role in this energy balance.

Table 59: Input-output balance of electrical energy and system expansion

Scenario	**Ref**		**Comp_Nat**		**Comp_Tech**		**Vac_Nat**		**Vac_Tech**	
Process	[MJ/(pe*a)]	[%]	[MJ/(pe*a)]	[%]	[MJ/(pe*a)]	[%]	[MJ/(pe*a)]	[%]	[MJ/(pe*a)]	[%]
INPUT	**162.9**	**100**	**102.4**	**100**	**160.8**	**100**	**211.4**	**100**	**249.4**	**100**
Drinking water supply	66.4	41	66.4	65	66.4	41	56.0	26	56.0	22
Wastewater treatment	85.0	52	7.2	7	65.8	41	5.7	3	45.8	18
Pumping/Vacuum	0	0	6.1	6	10.0	6	93.0	44	93.8	38
Composting	8.8	5	22.7	22	18.6	12	29.9	14	27.2	11
Digestion	0	0	0	0	0	0	26.8	13	26.6	11
Flocculant	2.7	2	0	0	0	0	0	0	0	0
OUTPUT	**57.3**	**100**	**15.7**	**100**	**17.8**	**100**	**163.5**	**100**	**155.1**	**100**
Sewage gas/ biogas	43.5	76	0	0	0	0	147.2	90	137.4	89
Sludge incineration	2.0	4	0	0	2.7	15	0	0	2.2	1
Biowaste incineration	11.2	20	11.2	71	11.2	63	11.2	7	11.2	7
Feedstock recovery*	0.6	0	4.5	29	3.9	22	5.1	3	4.3	3
Net energy demand [MJ/(pe*a)]**	**105.5**		**86.7**		**143.1**		**47.8**		**94.3**	
System expansion process***										
in MJ/(pe*a)	**106.2**		**147.8**		**145.7**		**0**		**8.4**	
in kWh/(pe*a)	**29.5**		**41.0**		**40.5**		**0**		**2.3**	

* incineration of plastic materials from construction
** input minus output
*** related to scenario with maximum output (Vac_Nat)
remark: values of zero can originate from rounding

In relation to the scenario with the maximum electricity output (Vac_Nat), the other scenarios are expanded with the supply of electrical energy in the respective amount. Depending on the scenario, 2 - 41 kWh/(pe*a) have to be supplied by conventional power plants. These figures are in the same order of magnitude than the net energy demand (input minus output) of conventional and composting systems. In other words, conventional and composting systems have to additionally generate at least their own net energy demand as electrical energy to be comparable to the vacuum scenario.

It has to be noted that in scenarios with technical greywater treatment (SBR), anaerobic sludge stabilisation and biogas use is not considered. Hence, conventional wastewater treatment with energy recovery via sewage gas usage needs effectively less energy than technical greywater treatment in this calculation. How far this assumption is reflecting realistic conditions has to be further investigated. By intuition, the treatment of wastewater with smaller loads of nutrients (=greywater) should be more energy efficient than conventional wastewater treatment.

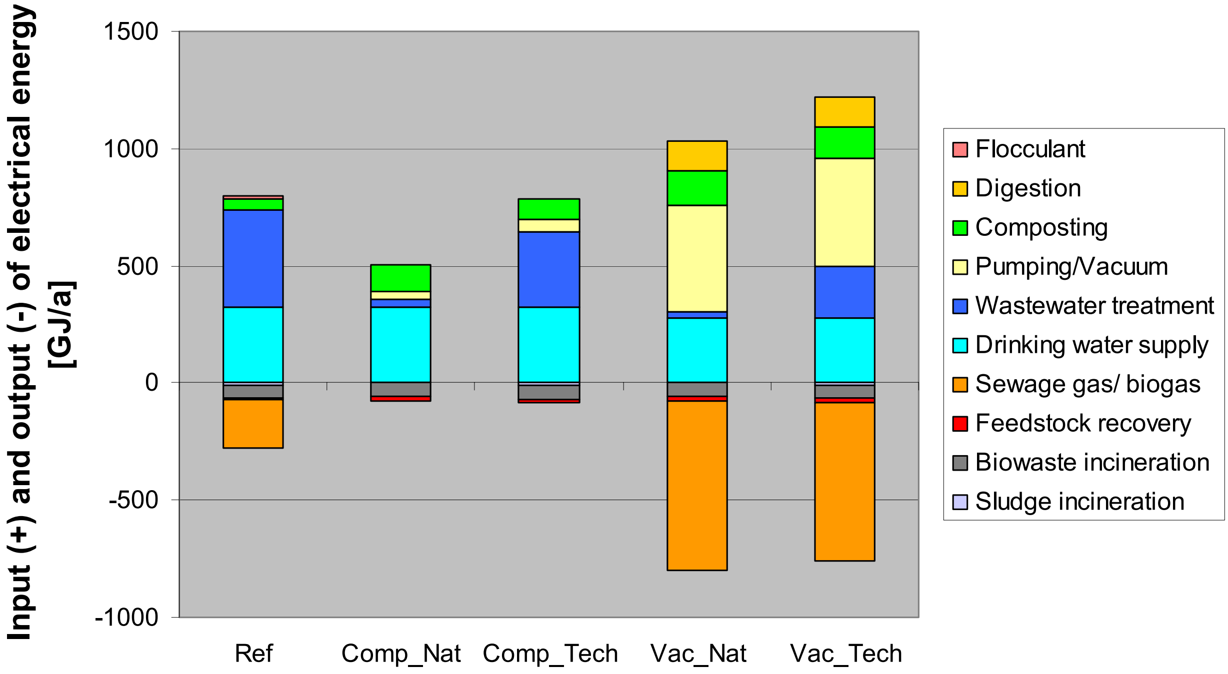


Figure 21: Allocation of operational electrical energy input and output

Thermal energy

In analogy to the electrical energy balance, a thermal energy balance is set up (Table 60). Energy in thermal form (heat) cannot be easily transferred from the producer to the user without considerable losses during transport and transformation. Thus, the substitution of thermal energy demand by a heat surplus at some place in the system may not be feasible. However, most of the thermal energy is produced and required in the digestion process (sludge stabilisation or faeces digestion). For the remaining fraction and the calculated system expansion, it is assumed that the thermal energy can be transferred and used without major losses. The influence of the thermal energy supply on the system comparison is very small, so the simplification of this balance does not affect the results of this study.

Table 60: Input-Output balance of thermal energy for each scenario

Scenario	Ref	Comp_Nat	Comp_Tech	Vac_Nat	Vac_Tech
Process			[MJ/(pe*a)]		
INPUT	**77.1**	**0**	**0**	**245.3**	**247.6**
Digestor	77.1	0	0	0	0
Hygienisation of faeces	0	0	0	245.3	247.6
OUTPUT	**176.2**	**82.2**	**102.2**	**346.3**	**342.1**
Sewage gas/ biogas	77.7	0	0	262.3	242.3
Sludge incineration	26.4	0	21.7	0	18.0
Biowaste incineration	70.3	70.3	70.3	70.3	70.3
Feedstock recovery*	1.8	11.9	10.2	13.7	11.4
Effective energy output **	99.1	82.2	102.2	101.0	94.5
System expansion	3.1	10.0	0	1.2	7.7

* incineration of plastic materials from construction
** input minus output

4.2 Selected inventory results

Inventory results from Life Cycle Inventory (LCI) are presented for nutrient and heavy metal flows in the different scenarios.

4.2.1 Nutrients

The distribution of nutrients from input flows to the different environmental compartments (water, air, soil) or to waste disposal (sludge ash) shows the recycling of nutrients in alternative sanitation concepts (Table 61). 58-64% of input-N, 61-64% of input-P, and 42-55% of input-K are recycled to agriculture in this calculation (plant-availability not accounted), so the major part of the nutrients in the different wastewater fractions can be reused as fertilizer (Figure 22). In the conventional scenario, nitrogen is mostly emitted to air (denitrification), whereas phosphorus is bound in sludge (80%) and potassium is emitted in surface waters (67%). The small amounts of recycled nutrients in the conventional scenario derive from the composting and agricultural application of biowaste. An agricultural application of sewage sludge for fertilizing purposes is not considered within this LCA study.

Table 61: Nutrient flows for each scenario to water (w), air (a), soil (s), and waste

Scenario	**Ref**		**Comp_Nat**		**Comp_Tech**		**Vac_Nat**		**Vac_Tech**	
Nutrient*	[kg/a]	[%]	[kg/a]	[%]	[kg/a]	[%]	[kg/a]	[%]	[kg/a]	[%]
N (w)	4076	15	3449	13	1762	7	1735	6	1134	4
N (a)	15540	58	5914	22	6152	23	6195	23	6284	23
N (s)	**1395	5	15696	58	15741	58	17104	64	17129	64
N to waste	5899	22	1851	7	3247	12	1876	7	2363	9
P (w)	147	3	100	2	141	3	100	2	112	2
P (s)	**835	17	3092	61	3063	61	3212	64	3177	63
P to waste	4048	80	1838	37	1825	36	1718	34	1741	35
K (w)	8995	67	4409	33	4507	33	6014	44	6063	45
K (s)	**2350	17	7416	55	7318	55	5811	43	5762	42
K to waste	2187	16	1707	13	1707	13	1707	13	1707	13

* inputs: 26910 kg N, 5030 kg P, 13532 kg K
** from biowaste compost

Water emissions of N, P, and K can be substantially reduced in all alternative scenarios, but the treatment of faeces filtrate together with greywater in the composting scenarios minimizes this advantage for nitrogen in case of soil filter treatment (Comp_Nat) and for phosphorus in case of SBR plant (Comp_Tech). Beside the nutrients from faeces, the filtrate also contains the nutrients from urine which could not be separated (assumed separation efficiency: 75% of urine). This leads to a considerable nutrient load which is added to the greywater and which has to be removed in the subsequent treatment. Maximizing the urine separation efficiency has to be a major target to increase the recyclable nutrient fraction and disburden the greywater treatment process.

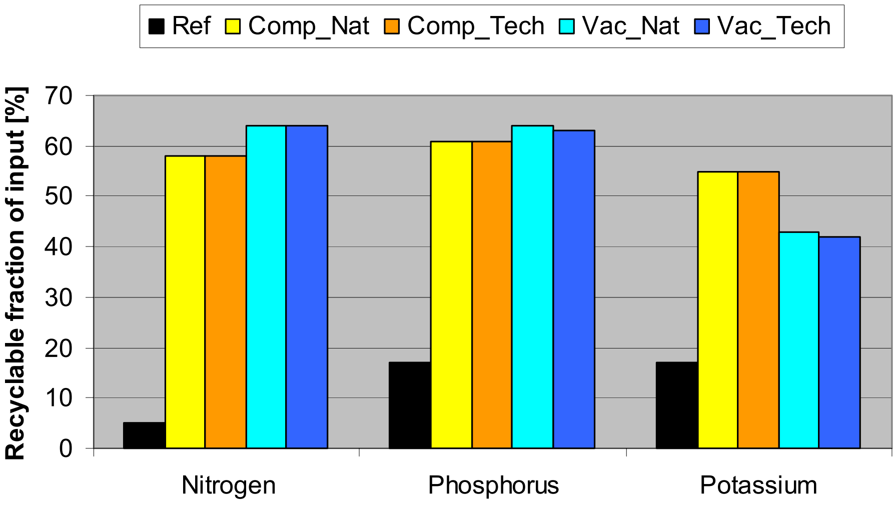


Figure 22: Fraction of nutrient input that can be effectively recycled as fertilizer

4.2.2 Heavy metals

Heavy metals (HM) are contained in each waste fraction of the input flows. While the HM content of urine and faeces is low, biowaste and loppings can contain considerable amounts of HM. Mineral fertilizer is also partially contaminated with HM (see chapter 3.3.3.2), and especially the application of industrial P fertilizer leads to increased loads of HM on agricultural soil. Table 62 lists the relevant loads of various heavy metals to agricultural soil and surface waters for the different scenarios.

Table 62: Total heavy metal loads to soil (s) and surface waters (w)

[kg/a]	WWTP	Ref	Comp_Nat	Comp_Tech	Vac_Nat	Vac_Tech
Cd (s)	0.06	0.33	0.11	0.11	0.08	0.08
Hg (s)	0.04	0.04	0.07	0.07	0.06	0.06
Cr (s)	1.22	4.89	1.50	1.47	1.11	1.09
Cu (s)	3.51	4.33	6.73	6.32	6.92	6.62
Ni (s)	0.69	1.41	1.19	1.15	0.75	0.73
Pb (s)	1.38	2.45	1.52	1.47	1.23	1.19
Zn (s)	22.50	29.28	42.06	41.14	33.89	33.31
Cd (w)	0.12	0.14	0.11	0.11	0.11	0.11
Hg (w)	0.01	0.03	0.01	0.01	0.01	0.01
Cr (w)	1.12	1.30	1.03	1.13	1.05	1.13
Cu (w)	6.67	6.81	7.64	6.26	6.72	5.53
Ni (w)	1.68	1.80	1.39	1.51	1.50	1.61
Pb (w)	1.19	1.34	1.10	1.25	1.15	1.28
Zn (w)	28.57	28.77	17.78	24.03	18.00	23.30

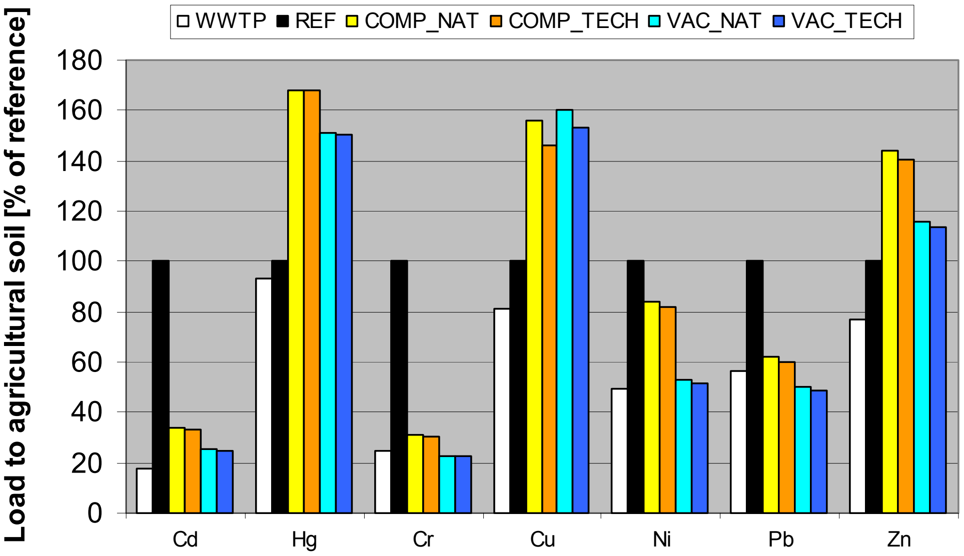


Figure 23: Changes in heavy metal loads to agricultural soil by SCST systems

By substituting the mineral fertilizer with secondary fertilizers from SCST systems, the HM on agricultural soil can be substantially decreased for certain potentially toxic substances (Cd, Cr, and Pb). However, urine and faeces contain micronutrients like Cu and Zn in elevated concentrations, so that the load of these metals to agricultural soil is increased by the application of secondary fertilizers. Uranium as a potentially toxic heavy metal is known to be contained in mineral P fertilizer, but it is neglected in this LCA study due to missing characterization factors for impact assessment. In general, the substitution of mineral fertilizer with secondary fertilizers from urine and faeces can decrease the loads of Cd and Cr up to 75% and of Pb up to 50% (Figure 23). The mercury (Hg) load may be increased, but the total mercury load is very small in all scenarios and does not significantly influence the ecological comparison.

The heavy metal loads to surface waters are not differing largely between the scenarios. In conventional wastewater treatment, most HM are reliably eliminated from the wastewater and transferred to the sewage sludge. Hence, the lower HM content of greywater does not lead to a significant HM load reduction to surface waters in the alternative scenarios (Table 62). However, the HM removal from the less polluted greywater can probably be efficiently reached even in a planted soil filter with post-precipitation. The LCI assumptions for the HM removal efficiency of the soil filter system have to be proved by long-term studies.

4.3 Impact assessment

After classification of the input and output flows to various impact categories, characterization of the substance and energy flows by specific characterization factors leads to the calculation of indicators. This section presents the results of the LCIA for the different indicators in detail, and relevant contributions to the single indicators are further discussed. The indicators are then weighted in relation to their relative significance through normalization. The normalized indicator values are finally summed up to form an overall eco-profile for the respective scenarios.

4.3.1 Impact categories

4.3.1.1 Energy demand

The indicator "cumulated energy demand" is a basic indicator for the cumulated demand for energy in various forms. This report includes the indicators "cumulated energy demand of fossil resources" (CED_{fossil}) and "cumulated energy demand of nuclear resources" ($CED_{nuclear}$). Renewable forms of energy (e.g. solar, hydropower etc) are not considered due to their unlimited availability.

The results for CED_{fossil} are presented in Figure 24, subdivided into expenditures for system construction, system operation, and system expansion processes (mineral fertilizer and energy).

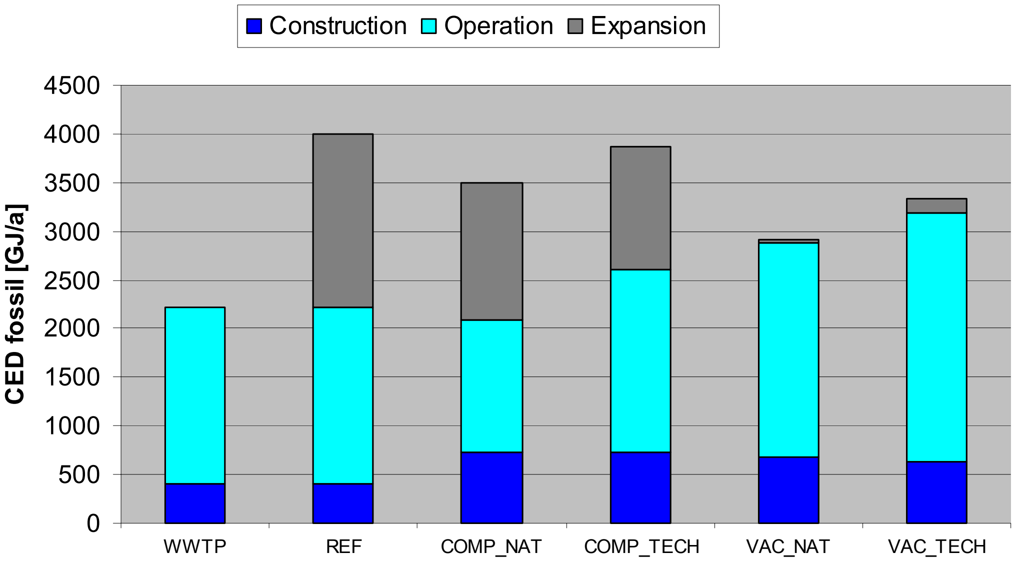


Figure 24: Cumulated energy demand of fossil resources (CED fossil)

CED_{fossil} of the conventional scenario (Ref) amounts to 4000 GJ/a. Thereof, the construction phase accounts for only 10%, whereas the production of additional fertilizer and energy for system expansion needs 45%, which is almost equal to the operational demand (45%). All alternative scenarios need more fossil resources for system construction and – except for scenario Comp_Nat – also for system operation, compared to the conventional case.

However, the large fossil energy demand for provision of mineral fertilizer and additional electricity generation in the conventional scenario offsets the energetic disadvantage of the alternative scenarios from construction and operation. Thus, alternative systems can reduce the total demand for fossil resources. The most efficient scenario is Vac_Nat, the scenario with vacuum drainage, subsequent digestion of faeces and energy-efficient greywater treatment in a soil filter, which can save up to 27% of the CED_{fossil} in this calculation.

A breakdown of the operational CED_{fossil} reveals that the energy savings from reduction of drinking water demand are marginal and do not play a major role in the energetic comparison (Figure 25). Decisive parts for the energetic performance of alternative systems are greywater treatment (SBR or soil filter) and faeces transport (vacuum system). The high energy demand of the vacuum system is compensated by the resulting energy production from co-digestion of faeces and biowaste. Here, the energy-rich organic content of biowaste contributes substantially to the energetic advantage of this system configuration. Faeces digestion without the addition of biowaste would yield considerably less biogas (-50%), so that the operation of the vacuum system would need more energy than is generated by the digestion process.

Urine separation, transport, and application need only a small fraction of fossil resources. However, the transport of large volumes of urine to the farms can consume considerable resources as fuel consumption is depending on the transport distance. The influence of urine transporting distance on the demand for fossil resources is therefore further investigated in sensitivity analysis (see chapter 4.4.2). The fraction of separable urine is assumed to 75% in this LCA study. Depending on the separation efficiency, more or less mineral fertilizer can be substituted, which can also affect the energetic comparison considerably. Hence, this influence is also investigated via sensitivity analysis.

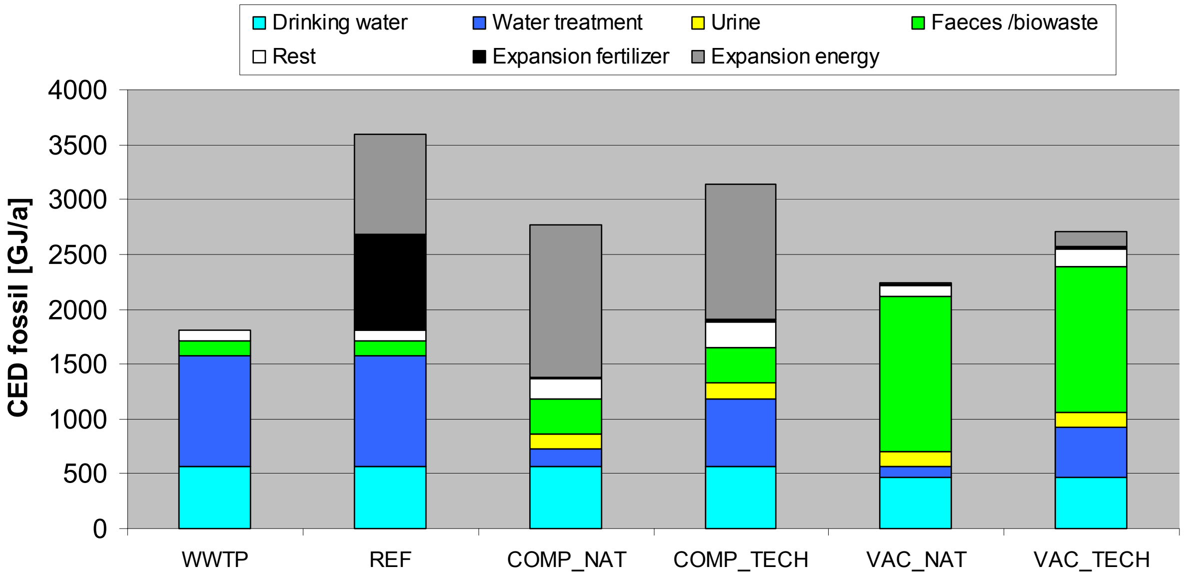


Figure 25: Detailed CED_{fossil} of operation and system expansion

A breakdown of CED_{fossil} for the construction phase shows the reason for the increased energy demand for constructing alternative systems. The multiple piping networks necessary for the drainage of the separated streams require more fossil resources (Figure 26). Together with the additional expenditures for the sanitary in-house installations, this leads to an increase of the constructional CED_{fossil} by 60 to 80%. Even though some fossil energy can be recovered by the thermal recycling of plastic pipes, the increased fossil energy demand for system construction is a clear disadvantage of alternative sanitation systems in the presented process configuration and boundary conditions.

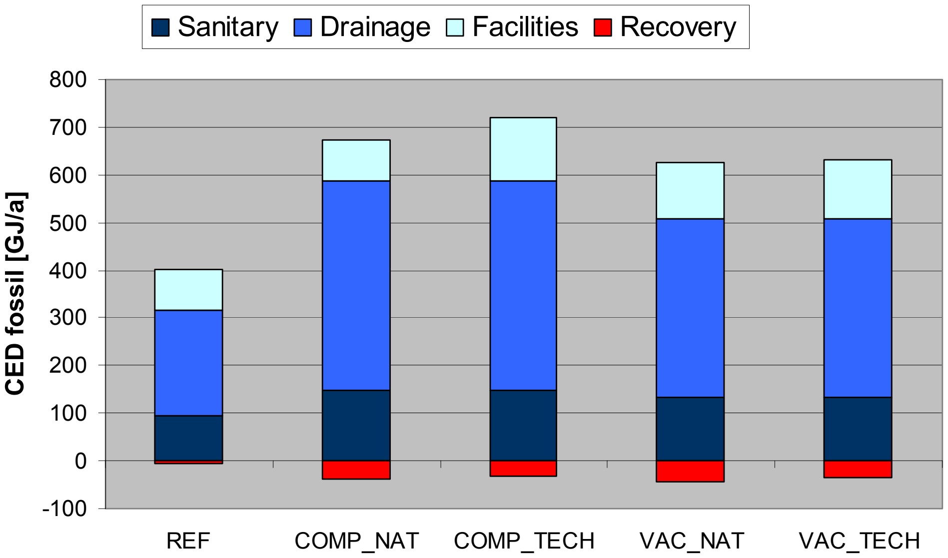


Figure 26: Detailed CED_{fossil} of construction

In all, it can be concluded from this calculation that alternative scenarios need more fossil resources for construction and also for the system operation than the conventional sanitation system, but this disadvantage is offset by the benefits of substitution of mineral fertilizer and energy supply. The contribution of the construction phase and the resulting energetic disadvantage for alternative systems may be further lowered if the assumed service life is increased (cf chapter 3.2.4). With energy-efficient alternative treatment processes like faeces digestion and natural greywater treatment, considerable amounts of fossil energy resources can be saved in total.

Concerning the demand for nuclear resources (which is closely related to the demand for electrical energy), the results are comparable to those of CED_{fossil}. The construction phase plays only a minor role, whereas the vacuum scenario needs more energy for operation than conventional and composting scenarios. This is again offset by the generated benefits of mineral fertilizer and electricity. Except scenario Comp_Tech, alternative scenarios use less nuclear resources than the conventional system (Figure 27).

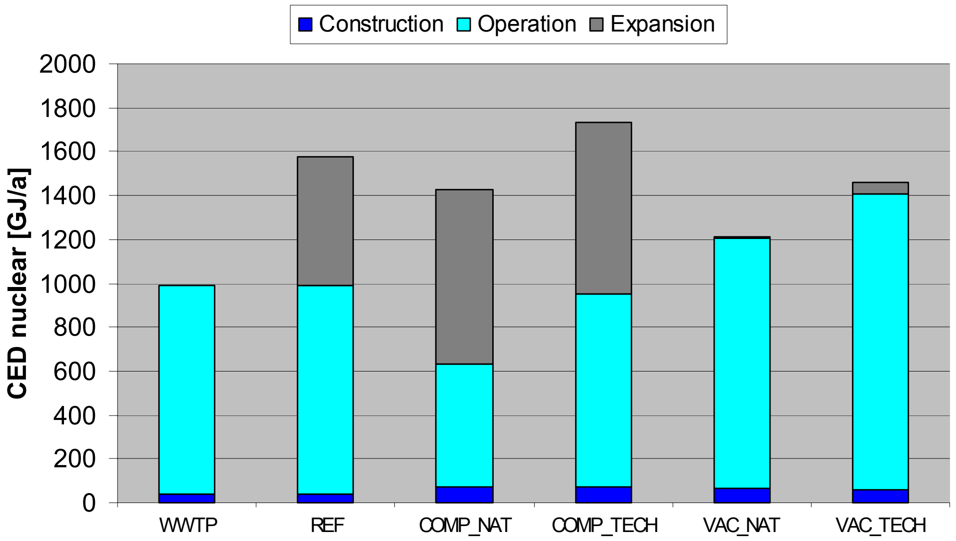


Figure 27: Cumulated energy demand of nuclear resources (CED nuclear)

4.3.1.2 Climate change

A well-known indicator for the rising problem of anthropogenic climate change is the global warming potential (GWP). Airborne emissions of fossil CO_2, N_2O, and CH_4 are contributing to this impact category. Figure 28 depicts the GWP for all scenarios, subdivided into contribution from construction, operation, and system expansion.

Similar to the cumulated energy demand, the construction phase plays only a minor role in the GWP comparison. Although alternative systems have a higher GWP in both constructional and operational phase, the benefits from system expansion offset these disadvantages, so that the conventional system has highest total GWP (600 Mg CO_2-eq/a). Depending on greywater treatment, composting scenarios can decrease GWP by 3 – 8%, whereas vacuum scenarios lower the GWP by 22 – 29%.

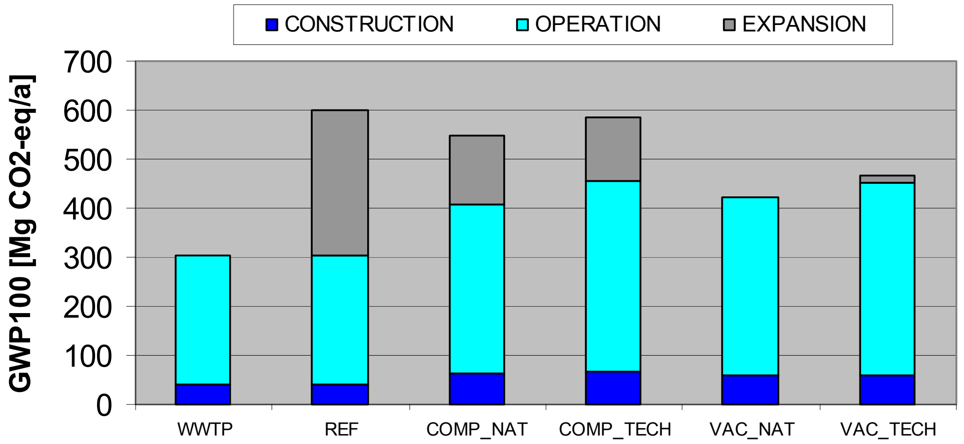


Figure 28 : Global warming potential (GWP)

GWP is closely linked to the power generation from fossil resources due to the emitted fossil CO_2. Other contributing processes are airborne emissions of N_2O and CH_4 from wastewater treatment, urine application and composting (Figure 29).

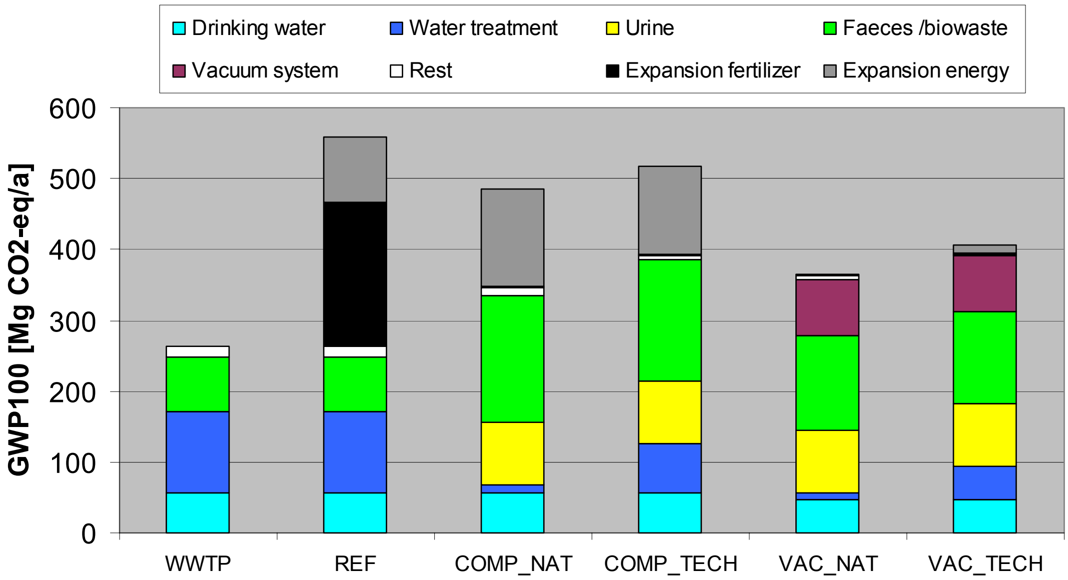


Figure 29: Detailed GWP of operation and system expansion

For all types of fertilizer (mineral or secondary), N_2O emissions during application are assumed to be equal (1.25% of nitrogen content). However, the plant availability of nitrogen differs considerably between fertilizers: compost has a tenfold lower nitrogen availability, and hence a worse emission balance of available N to emitted N_2O. This assumption has to be proved by further investigations of mineral and secondary fertilizer properties. The increased GWP of faeces and biowaste treatment in alternative scenarios arises partly from the unfavourable relation of $N_{available}/N_2O_{emitted}$ in compost (Table 63). If the composting process is

Table 63: Airborne emissions from composting and compost application

Scenario	Ref	Comp_Nat	Comp_Tech	Vac_Nat*	Vac_Tech*
Process	[kg/a]	[kg/a]	[kg/a]	[kg/a]	[kg/a]
Applied nitrogen from compost (plant-available)	140	322	312	490	481
N_2O emitted from composting process	89	224	220	33	33
N_2O emitted from application	29	73	72	102	101
Emission factor**	1.2	1.1	1.1	3.6	3.6

* composting of digester residual
** defined as N available/N_2O emitted (urine: 46.7, mineral fertilizer: 26.8 including energy processes)

encapsulated, the biofilter for off-gas cleaning generates additional N_2O from NH_3 oxidation (cf chapter 3.3.2.2). CH_4 emissions during the composting process further contribute to the poor emission balance of the composting process.

Nonetheless, the overall comparison of GWP between the systems reveals an advantage for the alternative systems. The energetic benefits of the alternative systems and the resulting reduction in fossil CO_2 emissions exceed the disadvantage of increased N_2O and CH_4 emissions from composting.

4.3.1.3 Depletion of abiotic resources

This impact category is related to the consumption of abiotic resources which are only available in limited quantities. Beside the fossil energy resources (lignite and hard coal, crude oil, natural gas), metal ores and other raw materials are included in this category.

The results of the indicator "abiotic resource depletion potential" (ADP) are presented in Figure 30. The indicator results are dominated by the consumption of fossil fuels for energy production. Thus, ADP results are similar to those of CED_{fossil}: the conventional scenario has the highest ADP (2230 kg Sb-eq/a), followed by the composting scenarios which reduce ADP by 2 – 12% compared to the reference. The digestion scenarios which have the lowest fossil energy demand can reduce ADP by 15 – 27% depending on the greywater treatment process.

The depletion of mineral phosphate resources is seen as a future problem, because the deposits of raw phosphates are limited and partially polluted with toxic heavy metals (particularly Cd). However, this aspect cannot be observed quantitatively in the indicator ADP. The ADP characterization factor for raw phosphate ($9.29*10^{-6}$) is smaller than those for

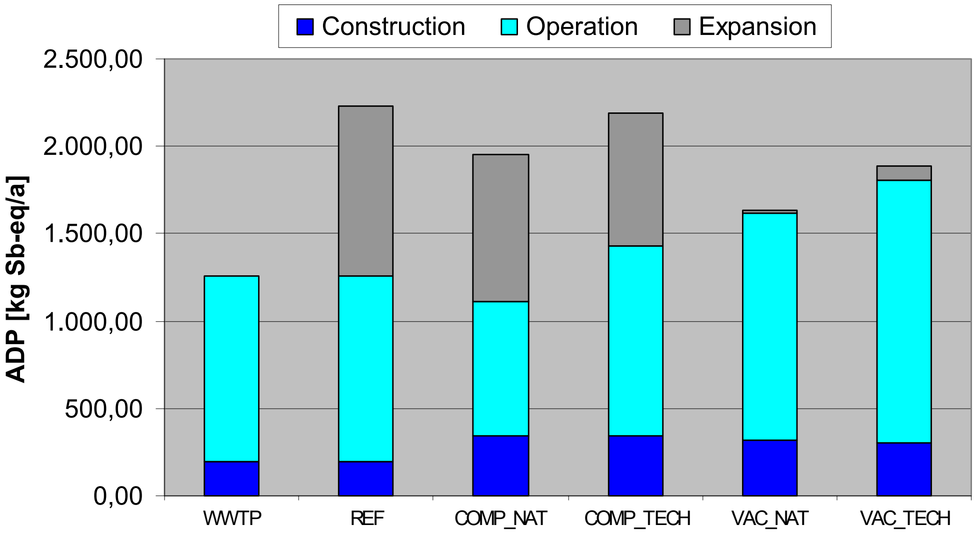


Figure 30: Abiotic resource depletion potential (ADP)

fossil fuels (1-2*10^{-2}) by around three orders of magnitude, so that the elevated demand for raw phosphates in the conventional system does not really influence the ADP indicator (< 0.1% of ADP). In other words, the advantage of recycling anthropogenic phosphorus in urine and faeces and thus substituting phosphorus from mineral deposits in alternative sanitation systems is not reflected in this indicator.

4.3.1.4 Acidification

Acidification is an environmental impact caused by airborne emissions of NH_3, NO_x, and SO_2. The wet deposition of these gases on soil and surface waters leads to a number of negative effects in all parts of the ecosystem (e.g. forest decline, fish mortality).

The results of the related indicator show that all alternative scenarios have a higher acidification potential (AP) than the reference scenario (Figure 31). Composting scenarios increase the AP of the reference scenario (2320 kg SO_2-eq/a) by over 50%, whereas digestion scenarios increase AP by 10%. The construction phase has only a negligible influence on AP (< 5% of total AP) in all scenarios.

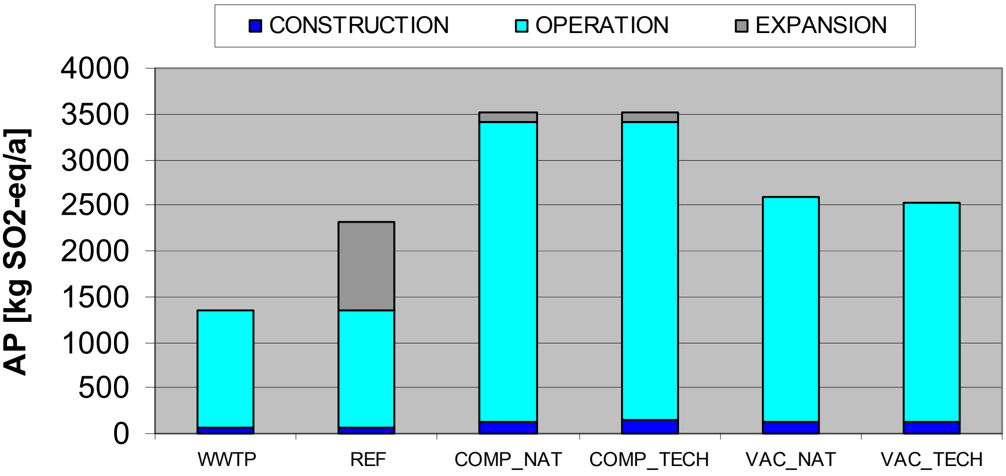


Figure 31: Acidification potential (AP)

The distribution of AP to the different processes reveals that emissions from energy supply are negligible in comparison to emissions from composting and fertilizer application (Figure 32). During the composting process, considerable amounts of the nitrogen content of the compost material are converted into NH_3. If this process is not encapsulated, NH_3 is emitted to the atmosphere. In the present process layout for composting scenarios, only the first and most intensive phase of composting is carried out in encapsulated boxes with off-gas cleaning, whereas the second phase is done in open piles. The digester residual of digestion scenarios is also stabilized aerobically in open piles. If these processes are fully encapsulated, NH_3 emissions and associated AP can be substantially reduced.

Another important contribution to AP results from the application of mineral and secondary fertilizers, especially urine. It is assumed that 6% of the nitrogen content in urine is emitted as

NH_3 after application. However, the use of mineral fertilizer also releases some NH_3 emissions, which is reflected by the proportional contribution of the expansion process in the conventional scenario.

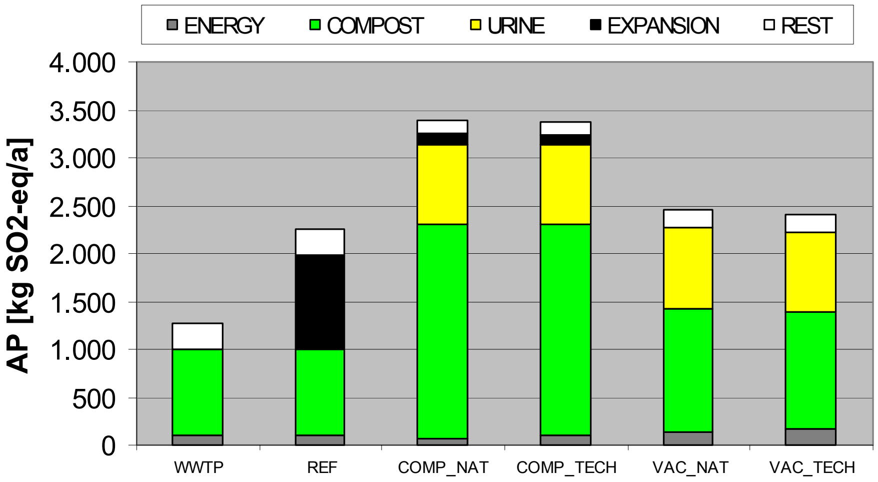


Figure 32: Detailed AP of operation and expansion

Both processes which mainly cause the increased AP in alternative scenarios (composting and urine application) can be modified to minimize NH_3 emissions. Composting processes can be laid out as a fully encapsulated process with off-gas cleaning. Urine can be applied with modern agricultural techniques (drag hose, liquid injection systems) to minimize nitrogen losses as NH_3. The effect of these minimization strategies on the AP indicator is further investigated in sensitivity analysis (see chapter 4.4.3).

4.3.1.5 Eutrophication

The emission of nutrients or degradable organic material to surface waters can trigger the eutrophication of these waters. Usually, one nutrient is limiting in an ecosystem and prevents excessive primary production and subsequent eutrophication. Whereas inland waters are mainly limited in phosphorus, marine waters are limited in nitrogen. Excess supply of those nutrients leads to eutrophication, whereas the degradation of organic material (as COD) can increase oxygen deficiency and further contributes to the problem.

Table 64 summarizes the aquatic emissions of nutrients and COD for the investigated scenarios. They occur mainly via the effluent from wastewater or greywater treatment. In the composting scenarios, greywater is mixed with faeces filtrate which contains considerable amounts of nutrients and causes elevated emissions of greywater treatment. In the digestion scenarios, the wastewater from dewatering of the digestion residual prior to its aerobic stabilisation is treated in a separate SBR plant before being discharged. Thus, the treated greywater in scenarios Vac_Nat and Vac_Tech contains less nutrients and COD than in composting scenarios.

Table 64: Emission loads and calculated effluent concentration from treatment of wastewater or greywater

Scenario	Ref	Comp_Nat	Comp_Tech	Vac_Nat	Vac_Tech
Emission loads [in g/(pe*a)]					
Nitrogen	833	705	360	355	232
Phosporus	30	20	29	20	23
COD*	2019	1877	1387	1324	985
Calculated effluent concentration [in mg/L]**					
Total N***	18	17	6	6	2
Total P	0.8	0.8	0.8	0.8	0.8
COD*	54	73	38	50	33

* conversion factor TOC/COD = 0.5

** from wastewater or greywater treatment, treated wastewater from dewatering of digester residual not included in scenarios Vac_Nat + Vac_Tech

*** NH_4-N+NO_3-N

The effluent in all scenarios has the same P concentration (0.8 mg/L) due to the chemical precipitation up to a target P concentration. However, the decreased effluent volume especially from soil filter treatment (evapotranspiration) leads to a calculational decrease in phosphorus loads to surface waters. To what extent this benefit can be realized under real conditions, this has to be further investigated. However, urine separation leads to a substantial decrease in P and N loads of the influent of wastewater treatment, which should improve the process performance and decrease the subsequent emissions via effluent.

The indicator chosen for this impact category is the eutrophication potential for P-limited watersheds (EP-limP). In contrast to other eutrophication indicators, this indicator only includes the emissions of phosphorus and COD into the calculation (Figure 33).

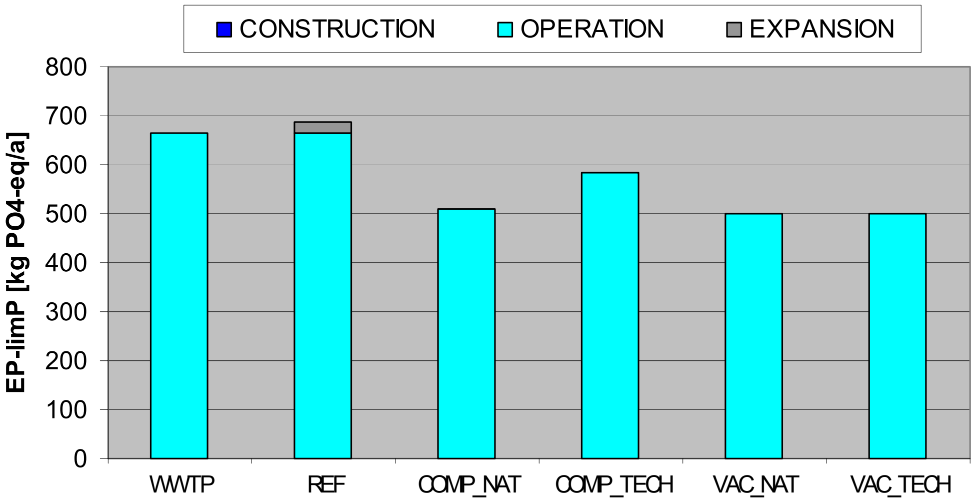


Figure 33: Eutrophication potential, P-limited (EP-limP)

The evaluation reflects the reduction of P and COD emissions in the alternative scenarios (cf. Table 64). Compared to the references system, the EP-limP indicator can be reduced by 15% in Comp_Tech and 25 – 27% in scenarios Comp_Nat, Vac_Nat, and Vac_Tech.

If aquatic and airborne nitrogen emissions are additionally accounted for in the evaluation of the eutrophication potential, the ranking of the alternative scenarios is different (Figure 34). Although all alternative scenarios still reduce EP in comparison to the reference system, the least reduction is reached with scenario Comp_Nat (- 9%) due to high airborne nitrogen emissions from composting and limited denitrification during the treatment of greywater and faeces filtrate in the soil filter. The other scenarios show a higher reduction in EP (33 – 52%) due to better denitrification (SBR in Comp_Tech) and less nitrogen loads of greywater (no faeces filtrate in Vac_Nat and Vac_Tech).

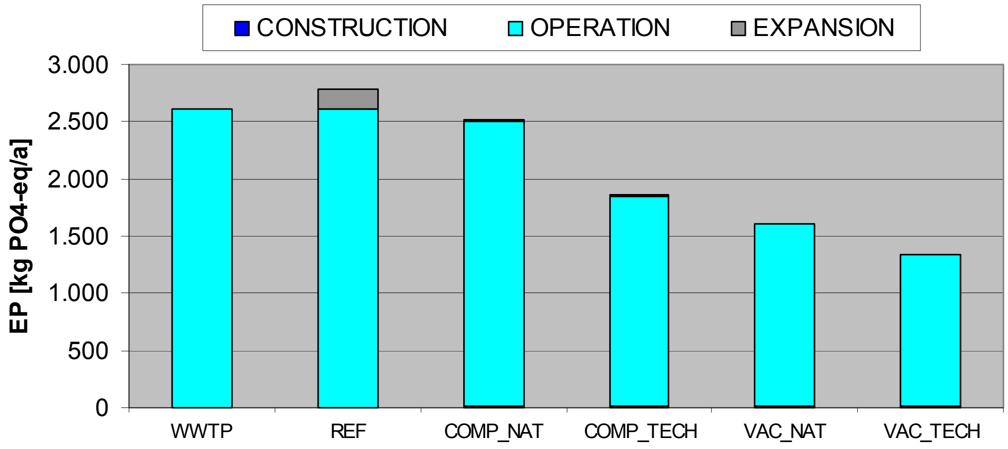


Figure 34: Eutrophication potential, including COD and nutrients N (in water and air) and P

The contribution of the different substances to the eutrophication potential is depicted in Figure 35. The difference between the two possible indicators (with and without nitrogen) can be clearly identified.

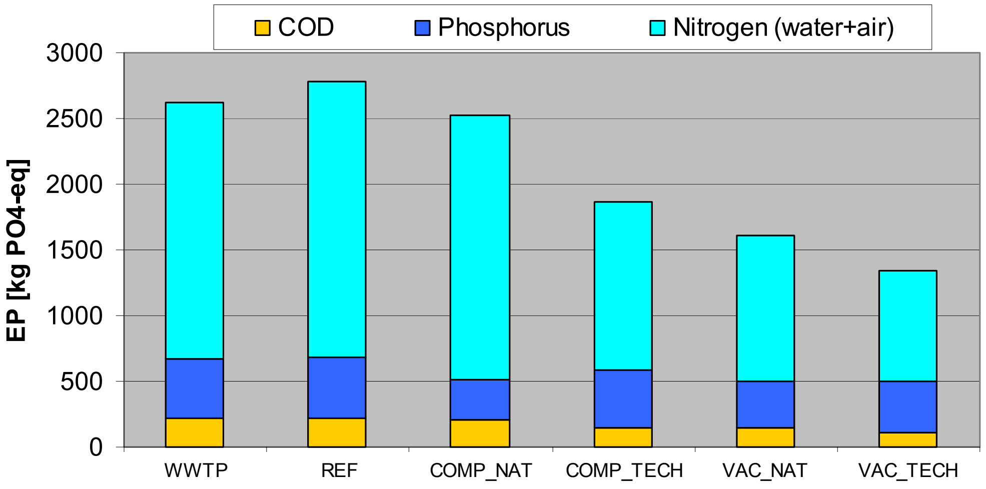


Figure 35: Contribution of N, P, and COD to eutrophication potential

In consultation with KWB, it is decided that the EP indicator for P-limited watersheds (EP-limP) should be used for the normalisation and further evaluation, because the surface waters in the Berlin-Brandenburg area and most of Germany are all limited in phosphorus. The proportional benefits for alternative systems from EP-limP reduction (minus 15 – 27%) are smaller than in the case of EP including nitrogen (minus 9 – 52%).

Hence, the decisive factor for the performance of the alternative scenarios in relation to the EP-limP is the efficiency of phosphorus removal in greywater treatment. For the two scenarios with natural greywater treatment in a soil filter, a downstream chemical precipitation stage is enhancing the P removal of these process configurations. However, disadvantages of this process upgrade are chemical demand, increased complexity of process control, and precipitation sludge handling. To quantify the effect of a precipitation stage on the EP indicator, a sensitivity analysis of P removal efficiency in the soil filter system is performed in chapter 4.4.4.

4.3.1.6 Toxicity potential

The toxicity potential is calculated for three different categories: aquatic and terrestrial ecotoxicity, and human toxicity. For both aquatic and terrestrial ecotoxicity, two different evaluation systems are presented. All indicators for ecotoxicity are mostly reflecting the amount of toxic heavy metals that are emitted in the related environmental compartments. The heavy metal loads to water and soil are listed in detail in chapter 4.2.2.

Aquatic ecotoxicity

The distribution of the freshwater aquatic ecotoxicity potential (FAETP) is presented in Figure 36. The differences between all scenarios are relatively small, and the alternative scenarios show a marginal reduction in FAETP (< 12%). Emissions from wastewater or greywater treatment contribute around 60 – 70% to the indicator, whereas the remainder is caused by soil emissions via compost or fertilizer.

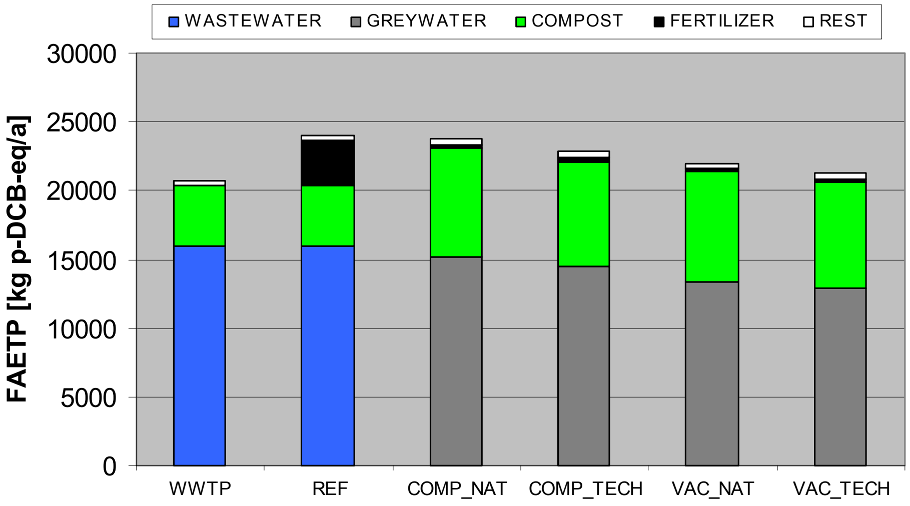


Figure 36: Freshwater aquatic ecotoxicity potential (FAETP)

Using another evaluation system for aquatic ecotoxicity (AET), alternative scenarios perform worse than the conventional scenario (plus 4 – 20%). Here, the heavy metal loads from fertilizer and especially compost application contribute more than 50% to the indicator in alternative scenarios. This is due to the elevated loads of Cu and Zn in compost, which have high AET characterization factors in this evaluation system.

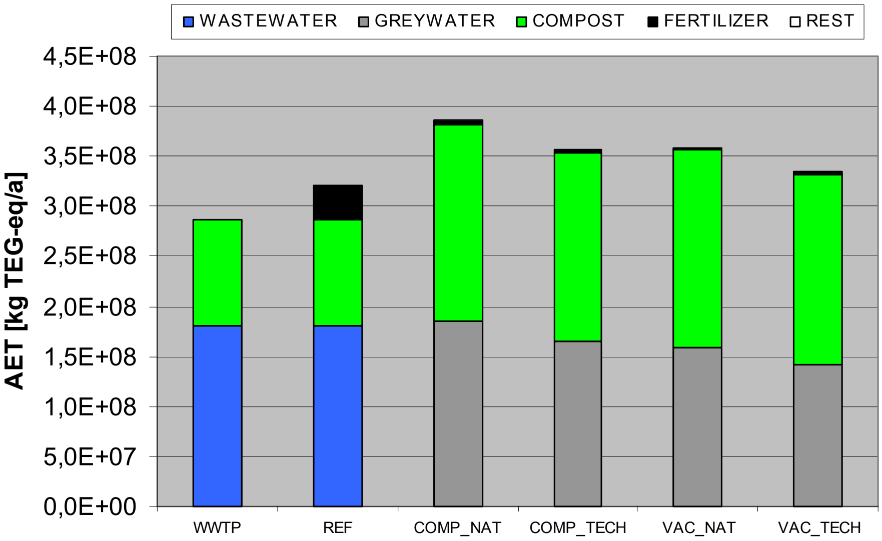


Figure 37: Aquatic ecotoxicity (AET)

Both indicators show that the emissions from water treatment are equal or slightly less in the alternative scenarios in comparison to the conventional system. However, a definitive statement about this impact category is difficult, as the two indicators show differing results.

Terrestrial ecotoxicity

The terrestrial ecotoxicity potential (TETP) is determined by the emissions of heavy metals to soil via the different types of fertilizer (Figure 38). The alternative scenarios have a significantly lower TETP than the conventional scenario (minus 56 – 64%). The heavy metal

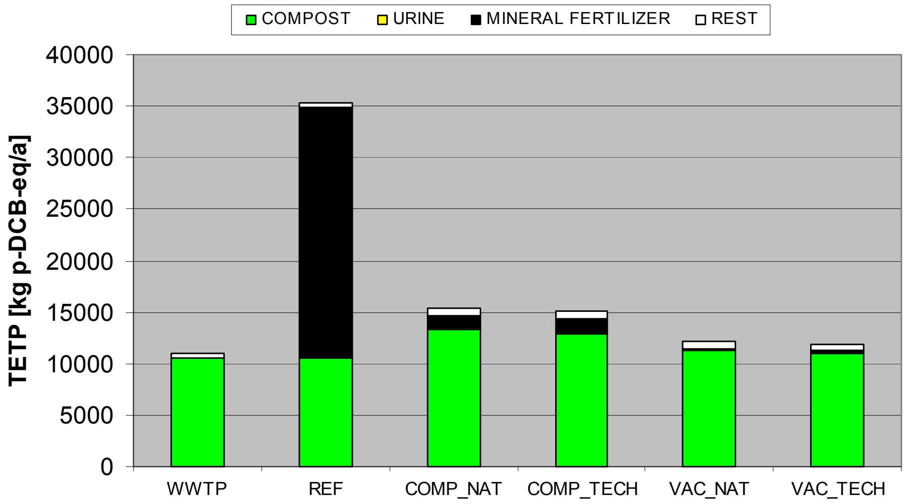


Figure 38: Terrestrial ecotoxicity potential (TETP)

content of mineral P-fertilizer is responsible for the high TETP of the conventional scenario. Its substitution with secondary fertilizers from urine and faeces lowers the soil loads of toxic substances such as Cd, Cr, and Pb substantially (cf. Figure 23).

However, the second indicator for terrestrial ecotoxicity (TET) reveals different results (Figure 39). Here, TET is increased in the alternative scenarios (plus 12 – 37%), which is mainly due to the high loads of Cu and especially Zn in SCST compost. These metals are essential trace elements for humans, and thus contained in compost from human faeces in elevated concentrations. Surprisingly, the high loads of Cu and Zn in SCST compost completely compensate the reduction of other toxic metals (Cd, Cr, Pb) in this indicator.

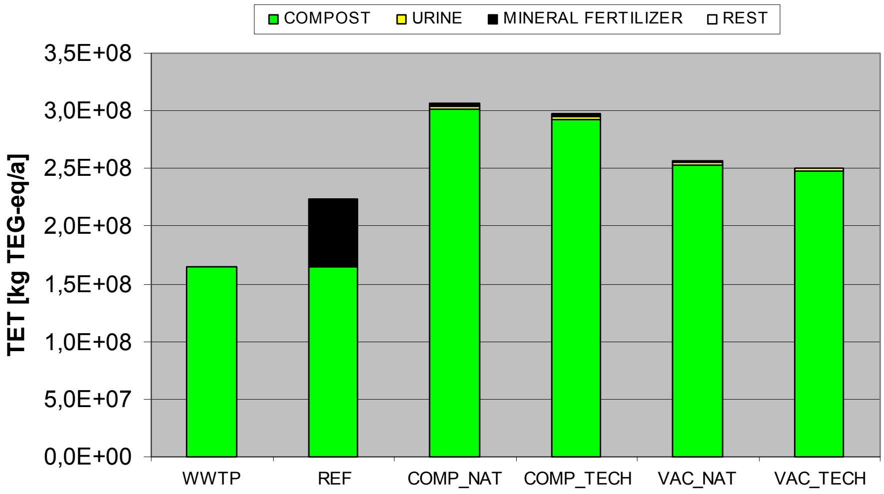


Figure 39: Terrestrial ecotoxicity (TET)

In all, the impact assessment for terrestrial and aquatic ecotoxicity mainly caused by heavy metal input into agricultural soil and water depends on the evaluation system. However, the toxicity evaluation is always affected with a great range of uncertainty due to the complex processes of pollutant speciation, fate, exposure, and effect in the ecosystem. For the following normalisation and further interpretation, the toxicity indicators originating from CML method are used. Hence, it has to be kept in mind that the results for these impact categories may vary considerably, if other evaluation methods are applied.

Human toxicity

The toxicity for humans is mainly determined via airborne emissions of heavy metals and other toxic substances (e.g. HF, dioxins), which are incorporated by humans through the respiratory system. The corresponding indicator for the human toxicity potential (HTP) shows that the construction phase causes a considerable proportion of HTP (Figure 40). This effect mostly derives from the emission of highly toxic dioxins in steel production. Industrial P-fertilizer production is responsible for another major contribution to HTP through process-specific HF emissions and again through the high heavy metal content (Cr, Cd, Pb, Ni). Overall, the alternative systems cause a reduction of 45 – 65% in HTP in comparison with the conventional system, mainly due to the substitution of industrial P-fertilizer.

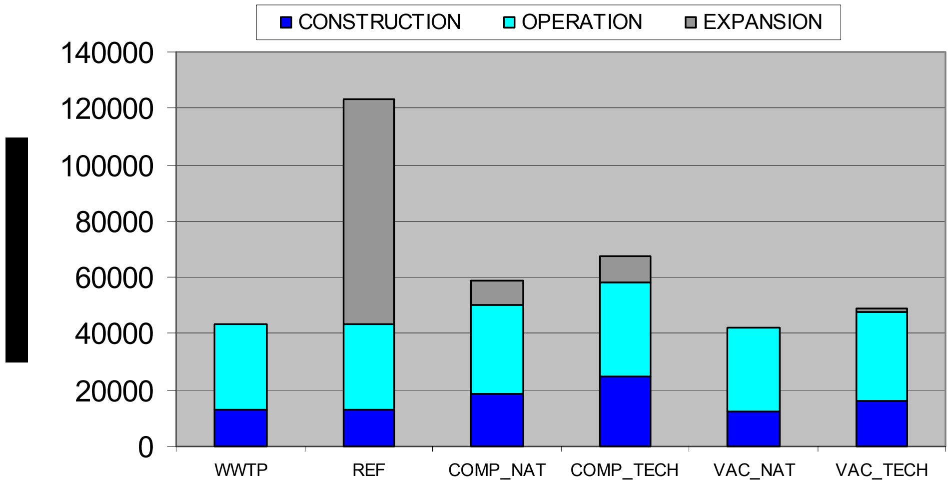


Figure 40: Human toxicity potential (HTP)

4.3.1.7 Summary of all indicators

Figure 41 summarizes the results for all LCIA indicators in proportional relation to the reference scenario. The alternative scenarios perform better in the majority of CML impact categories, except in acidification (AP) and the two additional ecotoxicity indicators (AET and TET) originating from another LCIA method. The exact data for all indicator results is listed in the annex.

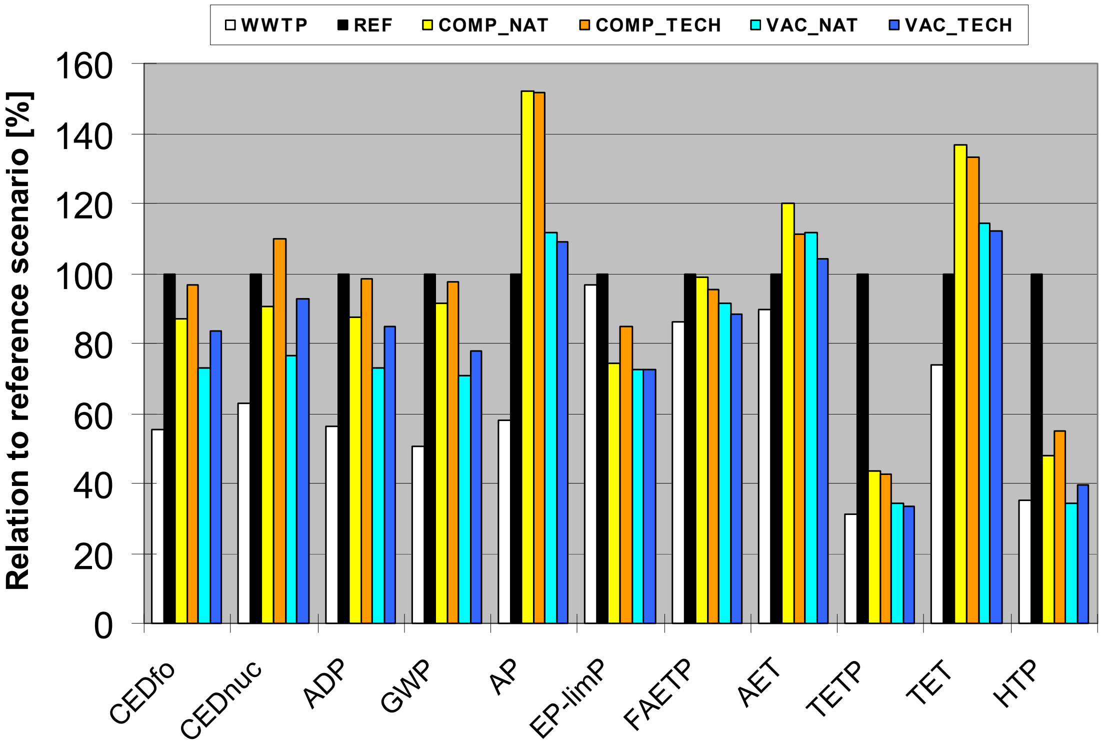


Figure 41: Overview of all LCIA indicators in relation to reference scenario

4.3.2 Normalisation

The LCIA indicators are normalised to German conditions to allow a comparison of the different indicators in relation to their share of the overall environmental impact of the respective category. The normalised indicators can thus be directly compared on an identical scale (Figure 42, see annex for numerical data). However, this is not an explicit weighting based on value choices (monetary values, policies, standards, expert panels etc).

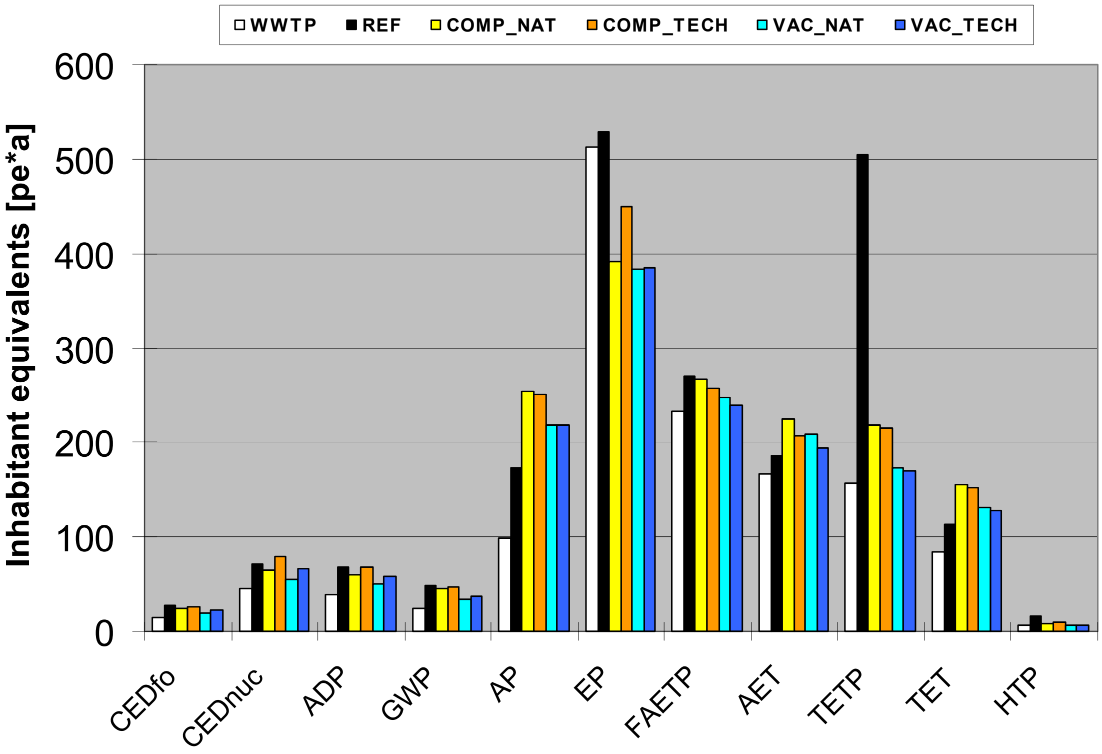


Figure 42: Overview of normalised LCIA indicators

While mostly energy-related indicators (CED, ADP, GWP) have relatively small relevance after normalisation (< 100 inhabitant equivalents (IEq)), emission-related indicators are more important. Certainly, indicators which are determined by aquatic emissions (EP, FAETP/AET) are to play a major role in evaluating sanitation systems. While nutrient emissions to surface waters are most important in this comparison via EP, aquatic emissions of heavy metals are less relevant (FAETP/AET). Another important indicator relates to the secondary function of fertilizer supply and the resulting input of heavy metals into agricultural soils (TETP/TET). Acidifying airborne emissions are fairly important (AP), while climate-active gases play a minor role (GWP). Finally, the potential for human toxicity (HTP) is the least important issue after normalisation.

The LCIA indicators can be divided into three groups: highly relevant indicators (> 250 IEq: EP, TETP), medium relevant indicators (100 – 250 IEq: AP, FAETP/AET, TET), and less relevant indicators (< 100 IEq: CED fossil and nuclear, ADP, GWP, HTP).

4.3.3 Eco-profiles

4.3.3.1 Eco-profiles related to German conditions

The normalised LCIA indicators can be summed up to form overall eco-profiles of the different sanitation scenarios (Figure 43). Thus, it is possible to draw general conclusions about the extent of the environmental impacts of the investigated systems. The contribution of each indicator to the eco-profiles can be easily compared, and a ranking of the different scenarios can be derived.

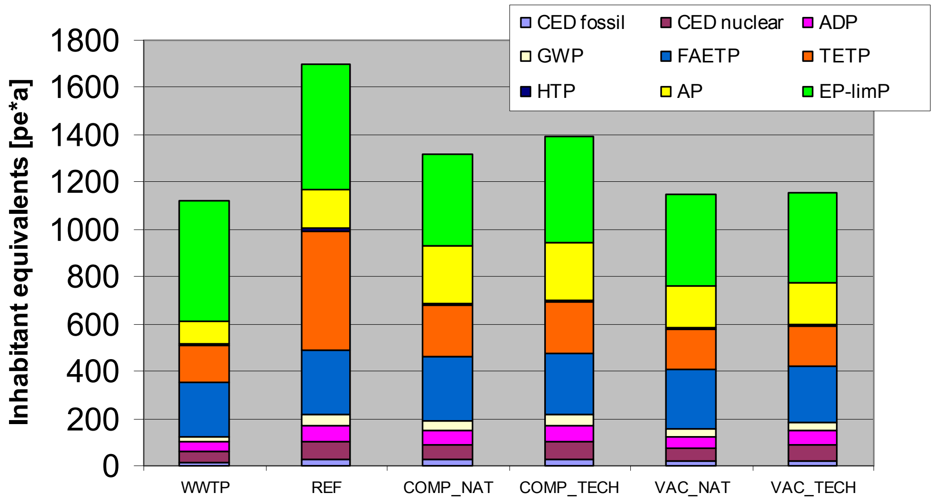


Figure 43: Eco-profiles of conventional and alternative sanitation systems

The eco-profiles show that the expanded reference system (REF) exhibits the largest environmental impact with around 1700 IEq. Composting scenarios have the second largest eco-profiles (-20% compared to REF), whereby the scenario with natural greywater treatment (Comp_Nat) has a small advantage over the technical greywater treatment (Comp_Tech). Both vacuum scenarios perform fairly equal and are those systems with the smallest eco-profiles (~ 1150 IEq), reducing the environmental impact by > 30% compared to the reference scenario.

The LCIA indicators EP, TETP, AP, and FAETP play the dominant role within the eco-profile comparison. While the alternative scenarios have advantages in eutrophication potential and especially in terrestrial ecotoxicity, they perform worse than the reference scenario concerning acidification. The aquatic ecotoxicity forms a significant fraction of the eco-profiles, but it is not decisive for the overall comparison, because all scenarios are performing almost equal on this indicator.

The contribution of the construction phase to the eco-profiles is determined to reveal the influence of this phase on the ecological comparison. Therefore, separate eco-profiles are calculated which only consider those environmental impacts caused by processes of system construction (material supply, construction energy etc). They are depicted in Figure 44, but these construction processes are already included in the eco-profiles shown above (Figure 43).

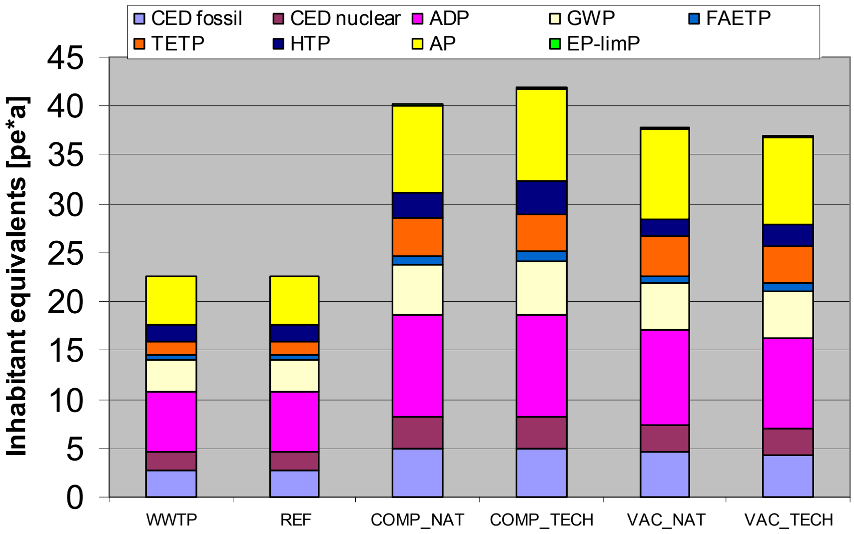


Figure 44: Eco-profiles of construction phase

The eco-profiles of the construction phase reflect the fact that has already been observed with the single indicators. Alternative systems with multiple piping networks need more energy for construction, so the energy-related indicators (CED, ADP, GWP) are higher than in the reference system. The same relation is valid for the emission-related indicators (TETP, HTP, AP), basically due to increased emissions from energy processes or material production. In all, the alternative systems increase the environmental impacts from the construction phase by 64 – 85% in this calculation.

However, the contribution of the construction phase (22 – 42 IEq) is generally small compared to the dimension of the overall eco-profiles (1150 – 1700 IEq). Thus, the increased expenditures during the construction phase are easily offset by the advantages of alternative systems during the operational phase. The construction phase constitutes a proportion of 1.5 – 4% of the overall environmental impacts, and this proportion would be further decreased if the assumed service life would be increased (cf chapter 3.2.4).

Alternative indicators

Some LCIA indicators can be defined with different characterization factors depending on the evaluation system. Within this study, alternative indicators are presented for aquatic and terrestrial ecotoxicity and eutrophication (for details cf chapter 2.2.1).

If these alternative indicators are used for normalisation and calculation of the eco-profiles, the system comparison can obtain different results (Figure 45). The alternative indicator for terrestrial ecotoxicity (TET) results in a disadvantage of alternative scenarios, while the same category revealed a considerable advantage for them while using the basic indicator (TETP). Now, the two composting scenarios have an equal or worse eco-profile than the reference system. However, the vacuum scenarios still represent the best performance on the ecological comparison due to the considerable reduction of the eutrophication potential. This advantage (basically due to the now accounted reduction of nitrogen emissions to surface waters) cannot be reached by the composting scenarios, because the co-treatment of faeces filtrate and greywater leads to substantial nitrogen emissions with the effluent.

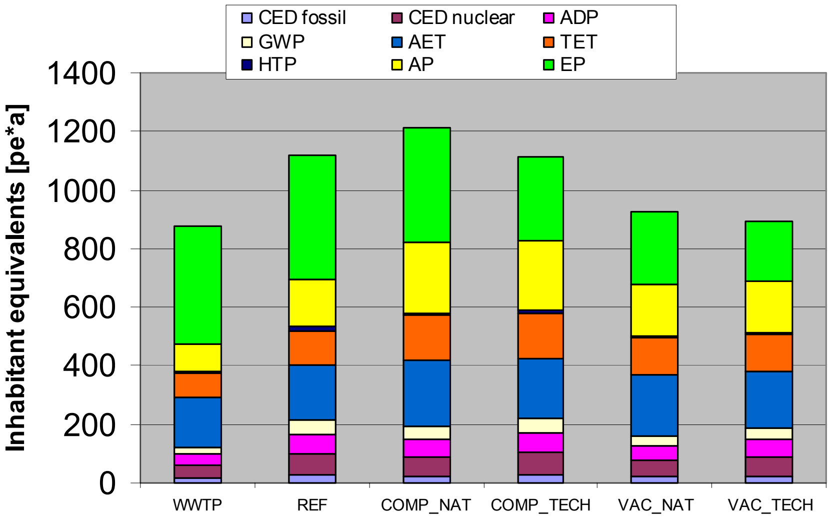


Figure 45: Eco-profiles with alternative LCIA indicators for ecotoxicity and eutrophication

This exemplary assessment of the consequences of indicator choice on the LCA results show that different indicators based on varying LCIA methods can lead to significant changes in the overall outcome of the LCA. Hence, the results of this LCA have to be seen in relation to the assumptions and definitions of the applied LCIA method and indicators.

4.3.3.2 Eco-profiles related to conditions of Western Europe

In this LCA study, the normalisation data is carefully generated from source data of relevant emissions and other related indicators for Germany in 2002. However, the calculation of the normalisation data may be affected with uncertainties and estimations for certain indicators. To reveal possible changes in the outcome and to conform with the chosen LCIA method completely, preset normalisation factors for Western Europe 1995 (from CML: Guinée et al., 2002) are employed (Figure 46). For this normalisation procedure, standard indicators for acidification and eutrophication are used. CED_{fossil} and $CED_{nuclear}$ are not included in the standard CML method, so they are normalized to German data instead.

The overall score of the eco-profiles is significantly reduced by the preset normalisation data. In contrast to 1150 – 1700 IEq in the “Germany case” presented above, the new eco-profiles are reduced to 290 – 620 IEq. Energy-related indicators (CED, ADP, GWP) play a more important role, whereas the contribution of the eutrophication indicator is considerably smaller than before. Although the comparison and the resulting ranking of the scenarios do not change with the preset normalisation data, the impact of the individual indicators on the overall comparison is clearly different. However, the general outcome of the ecological comparison is not significantly altered, so the influence of the German normalisation data on the LCA outcome can be characterized as minor compared to preset CML normalisation data.

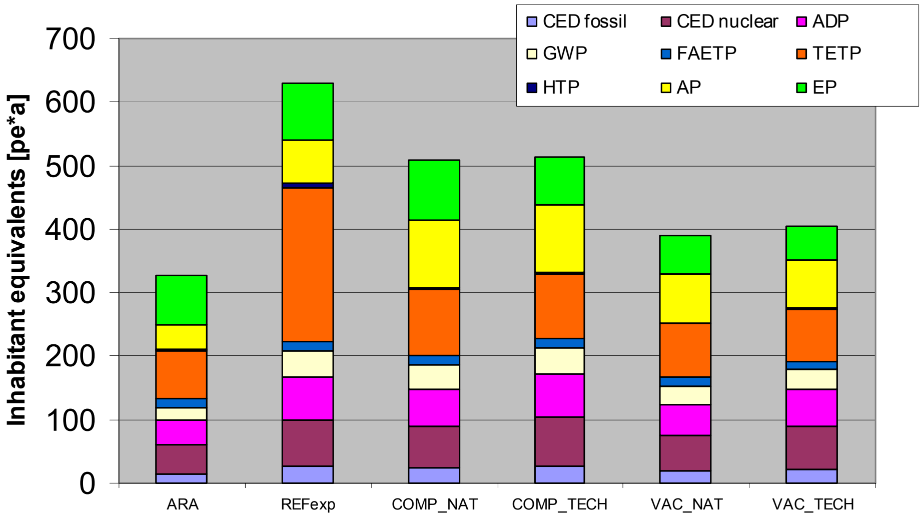


Figure 46: Eco-profiles for normalisation to Western Europe 1995

4.4 Sensitivity analysis

The results of the LCIA presented above depend on the process data which is deducted from pilot plants or literature. The stability of the LCIA results in relation to a variation of important system parameters is determined by sensitivity analysis. Therefore, a certain parameter is varied in an appropriate range, and the related LCIA indicators are monitored to quantify the influence of this parameter on the respective indicator. As the LCI data contains a huge amount of data, only selected parameters are investigated by sensitivity analysis.

4.4.1 Urine separation efficiency

The efficiency of urine separation determines the amount of urine that can effectively be separated and collected for further use as a fertilizer. In the LCI inventory, this parameter is set to 75% based on Swedish experiences (60-90%). However, in the SCST pilot plant in Berlin-Stahnsdorf, the distribution of nitrogen between separated urine and remaining brownwater has indicated that a considerable fraction of urine-N cannot be separated from brownwater and thus is lost for fertilizing purposes. Hence, the urine separation efficiency is an important system parameter, because it determines the amount of industrial nitrogen fertilizer that can be substituted by secondary fertilizers. The main LCIA indicators affected by the N-fertilizer production process are CED_{fossil} and GWP due to the considerable amount of energy that is necessary for the production ("Haber-Bosch-process").

In this sensitivity analysis, the urine separation efficiency is decreased from 75 to 60%. All related processes are adapted for the calculation (decreased industrial N-fertilizer production in reference scenario, enhanced nitrogen removal during faeces filtrate treatment in alternative scenarios, changes in transport loads etc).

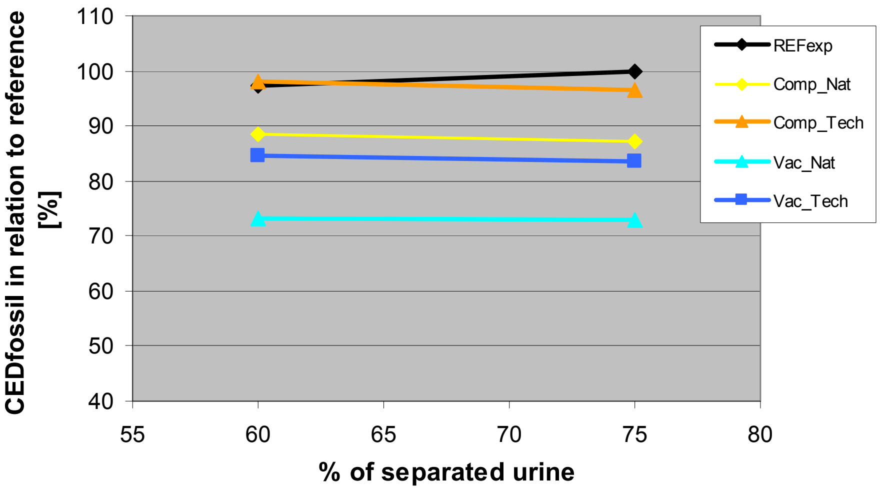


Figure 47: Variation of CED_{fossil} with decreasing urine separation efficiency

Figure 47 shows that the decrease of the urine separation efficiency to 60% leads to a reduction of the advantage in CED_{fossil} of the alternative scenarios. The ranking of the SCST scenarios does not change, but the alternative scenario with the highest CED_{fossil} (Comp_Tech) performs worse than the reference scenario at 60% urine separation efficiency. The same result can be observed with the GWP indicator (Figure 48). Again, the Comp_Tech scenario has a higher GWP than the reference scenario with decreased urine separation efficiency. It can be concluded from this sensitivity analysis that a decrease in the amount of separated urine to 60% can offset the energetic benefits of the composting system in comparison to the reference system. However, the energetic benefits of both vacuum scenarios are stable in a reasonable range of separation efficiency. In all, the urine separation efficiency is an important parameter for the energetic system comparison and should therefore be monitored closely.

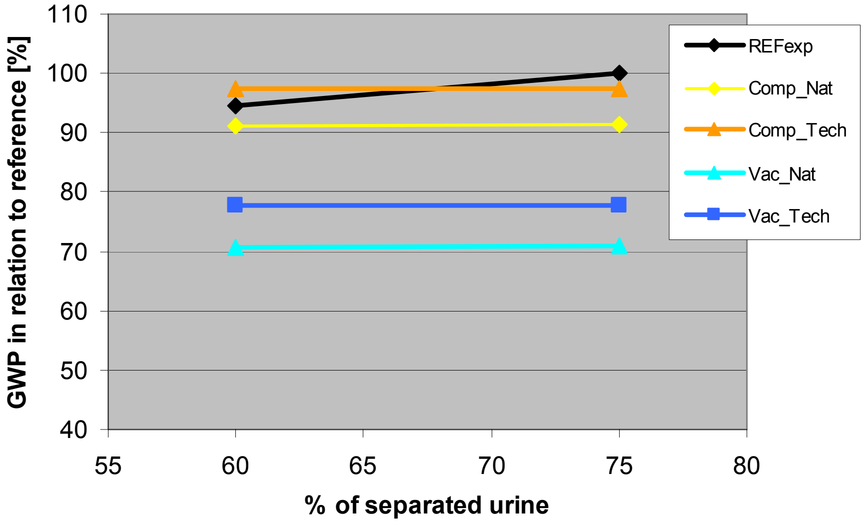


Figure 48: Variation of GWP with decreasing urine separation efficiency

4.4.2 Transport distance of urine and compost

The transport of large volumes of urine and compost over long distances can consume considerable amounts of truck fuel and cause respective emissions of CO_2. Hence, the transport distance of urine and compost from the settlement to the farms has an influence on CED_{fossil} and GWP. In this LCA study, this transport distance is estimated to be 10 km for urine and 20 km for compost. To quantify the influence of a longer transport on the relevant indicators, the transport distance is increased to 50 and 100 km for both urine and compost.

Figure 49 shows the variation of CED_{fossil} with the increasing transport distance. From this figure, threshold distances for each SCST scenario can be derived where the indicator CED_{fossil} is equal to that of the reference scenario (Comp_Nat: 127 km, Comp_Tech: 42 km, Vac_Nat: 268 km, Vac_Tech: 167 km). Above these distances, the energetic input for urine transport offsets the energetic benefits of the SCST systems. However, within reasonable transport distances for the secondary fertilizers (< 30 km) the energetic advantage of SCST systems is consistent.

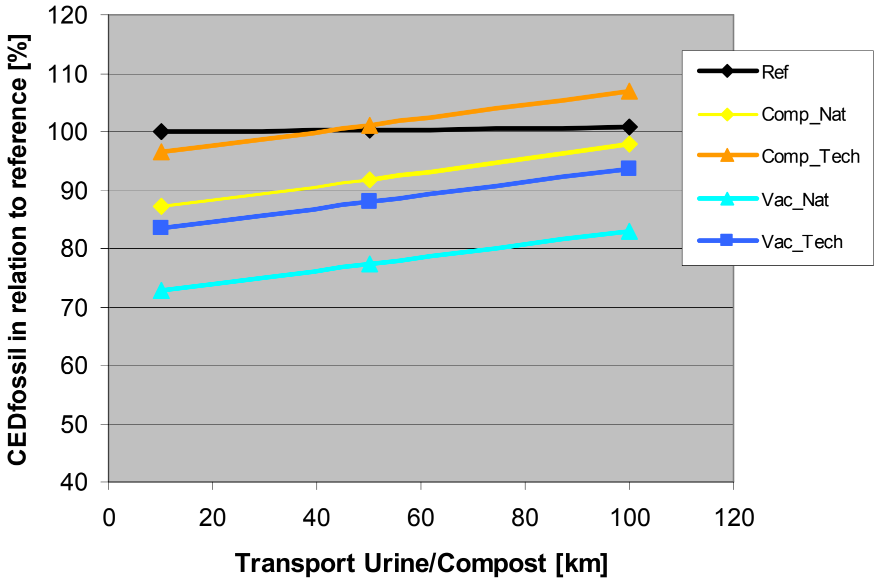


Figure 49: Variation of CED_{fossil} with increasing transport distance of urine and compost

Consequently, the GWP indicator reflects the same effects (Figure 50). Here, the threshold distances defined above amount to higher figures (Comp_Nat: 161 km, Comp_Tech: 54 km, Vac_Nat: 545 km, Vac_Tech: 418 km). Hence, the GWP indicator is not as sensitive to a variation in transport distance as the CED_{fossil}. In other words, if the CED_{fossil} offers advantages for SCST systems up to certain transport distances, their GWP are simultaneously lower than that of the reference scenario.

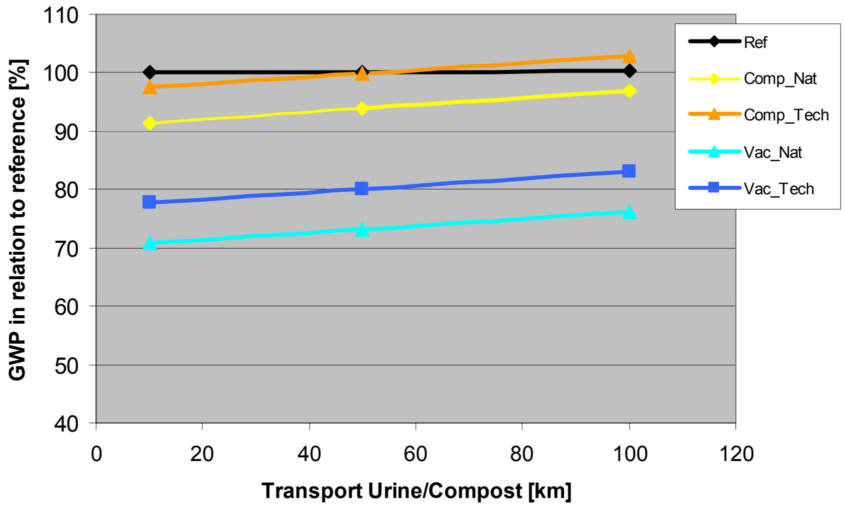


Figure 50: Variation of GWP with increasing transport distance of urine and compost

4.4.3 Ammonia emissions

The airborne emissions of ammonia (NH_3) are responsible for a major part of the acidification potential of the SCST systems. Especially ammonia emissions from the composting process are highly relevant beside emissions from urine application and others. Off-gas from the intensive stage of the composting process is cleaned in a biofilter, while off-gas from the second open composting stage is directly emitted to air. The efficiency of the biofilter in NH_3 reduction is preset to 60%, which is based on literature values.

In this sensitivity analysis, the influence of a variation of this NH_3 reduction potential of the biofilter is investigated. Furthermore, an encapsulation of the complete composting process with off-gas cleaning is considered. The relevant indicators under investigation are the acidification potential (AP) and the GWP due to partial conversion of NH_3 to climate-active N_2O in the biofilter.

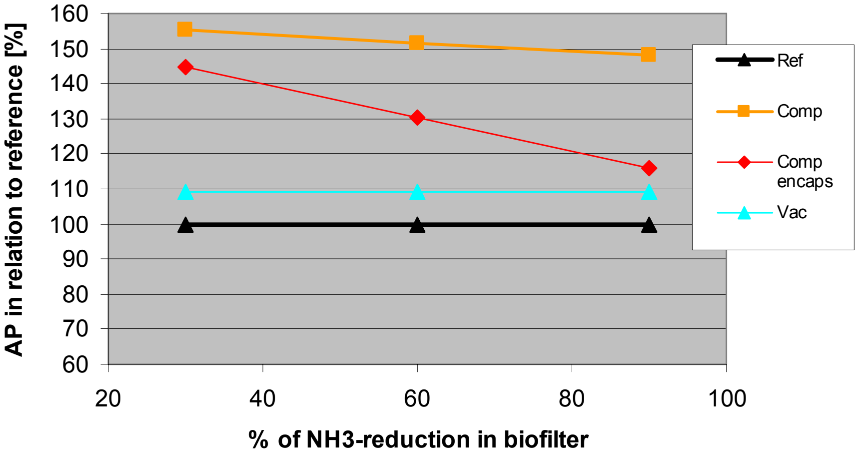


Figure 51: Variation of AP with NH_3 reduction in biofilter

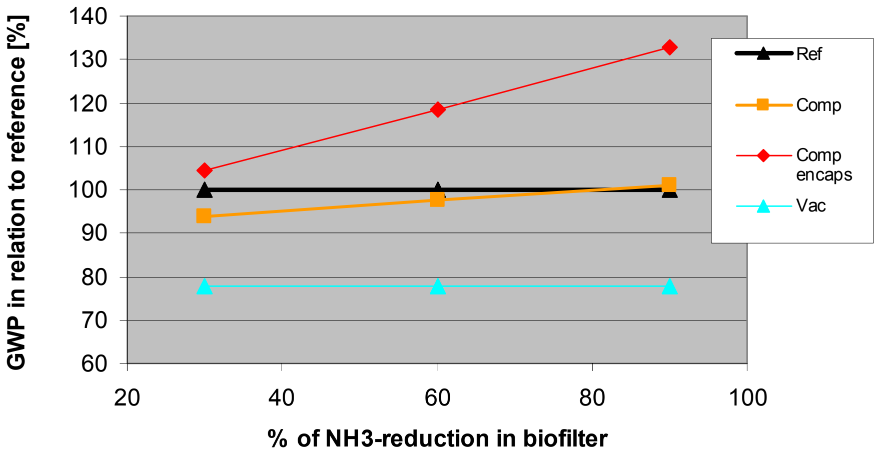


Figure 52: Variation of GWP with NH_3 reduction in biofilter

The increase of the NH_3 reduction in the biofilter has only a small impact on the composting scenario (Figure 51). Even if the efficiency rises from 60 to 90%, the AP indicator is still far above the respective values of reference and vacuum scenarios. Only with a fully encapsulated composting process, the AP can be substantially reduced. Still, the composting scenario exhibits a higher AP than the reference scenario in the best calculated case (+15%).

Furthermore, there is a trade-off with the GWP indicator due to the generation of N_2O in the biofilter. If more NH_3 is converted in the biofilter, more N_2O is emitted into the atmosphere (Figure 52). This conversion results in a worse GWP of the composting scenario compared to the reference system if the biofilter efficiency rises above 80% or if the composting process is fully encapsulated.

However, the acidification potential is an indicator with high impact on the system comparison after normalisation (cf Figure 43). Therefore, minimization strategies for ammonia emissions should be applied. A highly efficient biofilter without N_2O generation would reduce emissions of both harmful substances. Beside the modification of the composting process, NH_3 emissions during urine application should also be minimized by appropriate application techniques.

4.4.4 Phosphate removal in greywater treatment

The decisive parameter determining the eutrophication potential of SCST systems is the efficiency of phosphate removal from greywater during treatment in the soil filter or SBR. While the SBR system in scenarios with technical greywater treatment is usually equipped with a chemical precipitation, the soil filter system is originally designed without an additional precipitation stage. However, in the course of the study it became clear that long-term phosphate removal in the soil filter may not be sufficient to reach reasonable effluent concentrations. The influence of this system parameter on the eutrophication indicator is therefore investigated in sensitivity analysis.

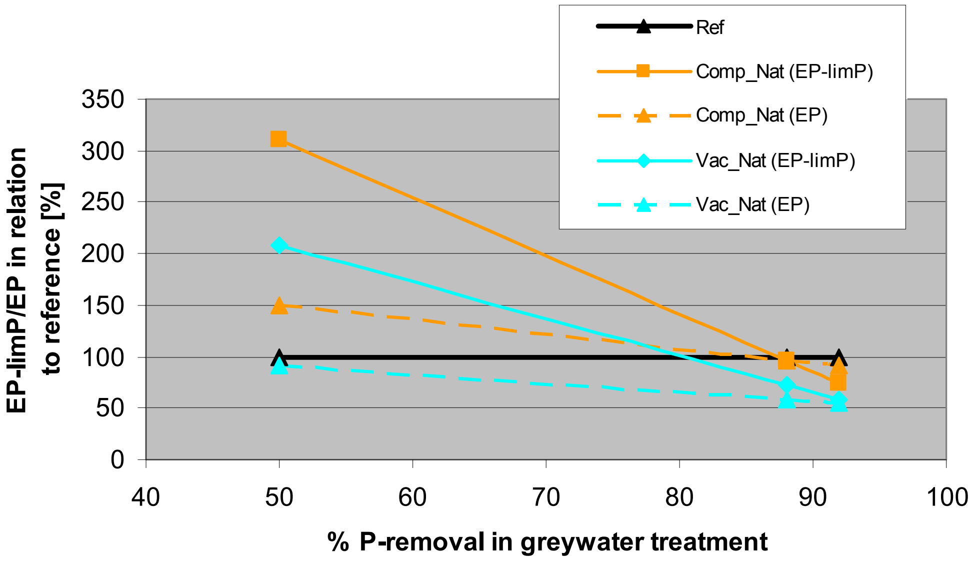


Figure 53: Variation of EP-limP and EP with phosphate removal efficiency of soil filter system

The variation of the indicators EP-limP (P only) and EP (N+P) with the efficiency of phosphate removal in the soil filter system is shown in Figure 53. In the LCI data, the long-term P removal of the soil filter system is estimated to 50%. An additional precipitation stage would increase the removal efficiency to around 90%.

Clearly, the effect of decreasing P removal on the eutrophication potential is more significant if faeces filtrate is treated together with greywater (Comp_Nat) due to increased P loads in the influent. Abandoning the option of an additional precipitation stage would increase the eutrophication potential to >200 % for scenario Vac_Nat or >300 % for scenario Comp_Nat. If nitrogen and phosphorus are accounted for eutrophication (indicator EP), this influence is smaller, but still significant for the composting scenario (+50% compared to reference).

Bearing in mind the great influence of the eutrophication indicator on the overall system comparison (cf Figure 43), the phosphate removal efficiency should not be smaller than 80% for the vacuum and 88% for the composting scenario. If these figures cannot be maintained in the long-term by the soil filter system alone, the provision of an additional precipitation stage is recommended. Otherwise, the increased eutrophication potential would seriously change the normalised eco-profiles of the SCST scenarios calculated in this LCA study.

In the KWB pilot plant in Berlin-Stahnsdorf, the P removal of the soil filter is increased by the application of iron sludge from waterworks into the filter bed. Thus, the reduction of phosphorus could be maintained at 75-82% during the first years of operation. However, the soil filter system was not loaded with its design capacity at the beginning of the investigation period.

4.5 Processes for urine treatment

Different treatment processes for source-separated urine are evaluated by a separate LCA assessment. For a better comparison, this LCA only includes the transport of the urine from the collection tanks in the settlement to the treatment facility, the treatment process itself, and the final transport of the fertilizer products to the farms. Some processes which are connected to the urine treatment have to be neglected due to the structure of the model: the supply of flocculant and the sludge disposal for the SBR treatment of depleted urine, and the energy for pumping of the urine. However, these processes constitute only a minor share of the environmental impacts.

Most of the environmental impacts from urine treatment are associated with the energy demand and related emissions for energy production. The relevant indicators CED_{fossil} and GWP are presented together with the normalised eco-profiles of each treatment process. Finally, different ways of nitrogen recycling are compared in relation to their specific energy demand.

4.5.1 Relevant LCIA indicators: CED_{fossil} and GWP

The relevant indicators for the ecological comparison of the different urine treatment processes are related to the energy demand. Figure 54 shows the results for CED_{fossil} and GWP for all investigated methods. Both indicators are strongly related with each other. The simple storage of urine consumes a negligible amount of fossil resources for transport purposes. The process of struvite precipitation and subsequent ion exchange with zeolithes needs more energy, and causes respective emissions of climate-active gases. Steam stripping in combination with evaporation or struvite precipitation has the highest energy demand and consequently the highest GWP.

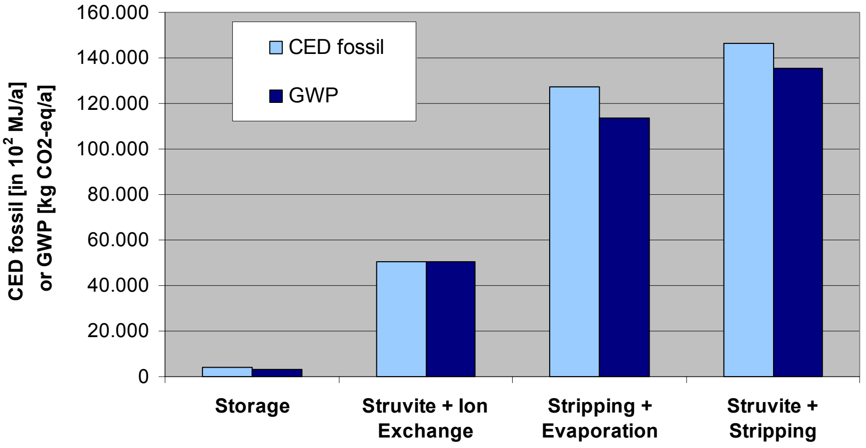


Figure 54: CED_{fossil} and GWP of different processes for urine treatment

The distribution of energy demand among the sub-processes of urine treatment methods shows that the ozonation process consumes a considerable part of the total energy demand of struvite precipitation and ion exchange. The supply of raw materials and the separation of the solid fertilizer need only small amounts of energy. A major source for the high energy demand of the stripping processes is the generation of steam which consumes high amounts of fossil energy sources and electricity. Due to the assumed energy savings which can be potentially realized by the process combination of stripping and evaporation, this process has slight energetic advantages compared to the combination of stripping and struvite production. However, the high energy demand of steam generation may completely offset the energetic advantages from nitrogen recovery and thus make this process energetically unfavourable. The production of struvite and N-loaded zeolithes is the preferable process in terms of energy demand.

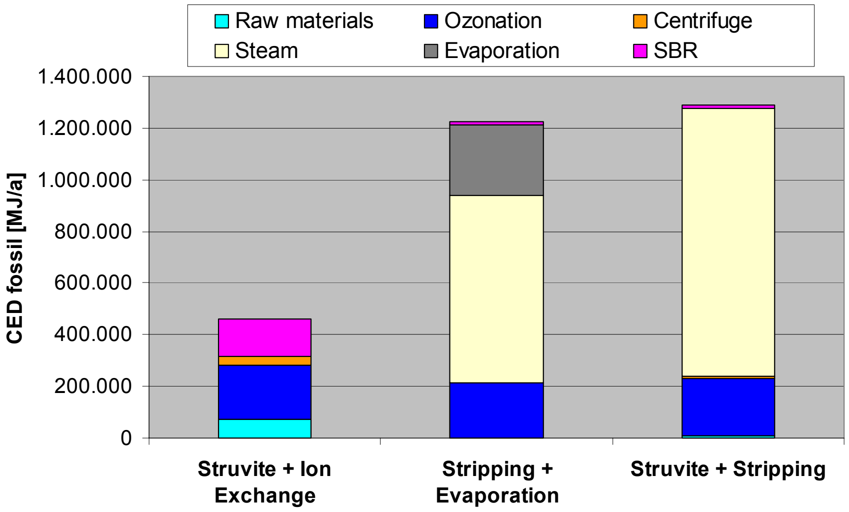


Figure 55: Distribution of $\text{CED}_{\text{fossil}}$ among sub-processes of urine treatment

4.5.2 Eco-profiles of processes for urine treatment

The normalised eco-profiles of all urine treatment processes are presented in Figure 56. The energy-related indicators ($\text{CED}_{\text{fossil}}$ and $\text{CED}_{\text{nuclear}}$, GWP, ADP) are decisive for the smaller eco-profile of struvite production and ion exchange. The acidification potential of the two process combinations with steam stripping results from potential losses of NH_3 to the atmosphere during stripping. If those losses can be minimized or completely avoided, the eco-profiles of the stripping processes would improve considerably.

Overall, the eco-profiles for urine treatment processes are small (40 – 100 IEq) in comparison to the total eco-profiles of the investigated sanitation scenarios (1150 – 1700 IEq, cf. Figure 43). Thus, it can be concluded that the implementation of any of the urine treatment processes investigated in this study would not offset the advantages of the alternative sanitation systems in this study. This is mostly due to the small relevance of energy-related indicators in the total eco-profiles of the sanitation systems.

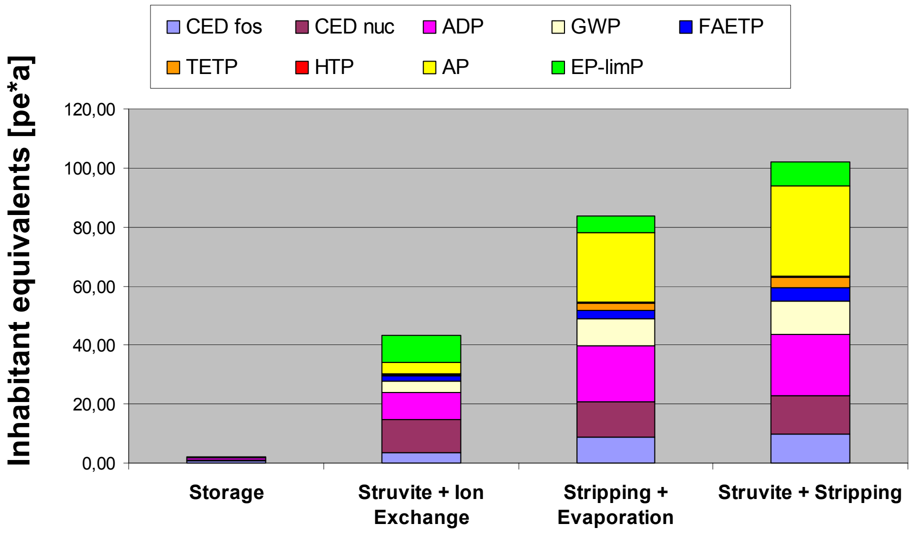


Figure 56: Eco-profiles of different processes for urine treatment

4.5.3 Energetic comparison of different ways of nitrogen recycling

Alternative systems are recycling a major fraction of the nitrogen from human excreta to agriculture. In the conventional system, this process is equivalent to the denitrification of nitrogen from the wastewater into elementary N_2 and the subsequent production of mineral nitrogen fertilizer by the fixation of N_2. Both processes need a considerable amount of energy. If urine treatment processes are implemented into alternative sanitation systems, these processes also consume energetic resources. By relating the required energy demand to the amount of nitrogen that is recycled, a specific energy demand for each process of nitrogen recycling can be calculated (Figure 57). The conventional way of denitrification and fertilizer production consumes 74 MJ/kg N, whereas separation and storage consumed only 3 MJ/kg N for transport. Struvite production and ion exchange needs 42 MJ/kg N, thus still being energetically favourable to the conventional system.

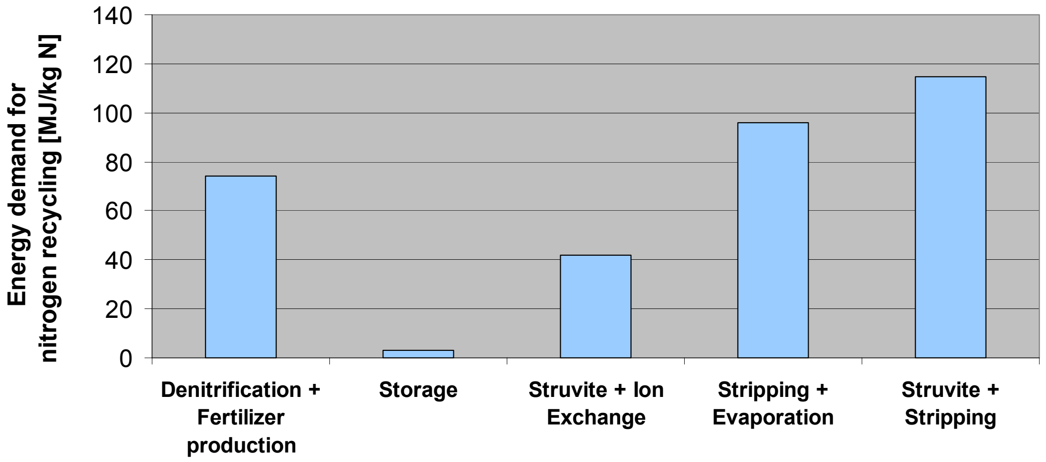


Figure 57: Specific energy demand for different ways of nitrogen recycling

Both urine treatment processes which involve steam stripping of urine-N need more energy than the conventional way of nitrogen recycling (96 MJ/kg N in combination with evaporation, 115 MJ/kg N in combination with struvite precipitation). Hence, these process combinations are not reasonable from an energetic viewpoint and have to be further optimized to decrease their energy demand.

5 Conclusions

The conventional sanitation system and several alternative sanitation systems have been evaluated in respect of their ecological performance with the methodology of Life Cycle Assessment. All relevant processes of system operation and system construction have been modelled in a substance flow model. The related data originates from pilot plants, literature, and qualified assumptions. The data quality of this prospective LCA study can be further improved, because some processes of alternative sanitation systems have not yet been realized in full-scale. However, the data quality seems good enough to allow a comprehensive analysis of the dimensions of the respective substance and energy flows and their contribution to the various impact categories. Hence, conclusions about the potential of ecological performance and sustainability of the alternative sanitation concepts can be drawn. Further improvement of data quality and updating of important system parameters with results from pilot studies etc. will strengthen these results. Whether the potential benefits of alternative sanitation concepts can be realized, that depends on the respective boundary conditions within a settlement and the technical implementation of the systems.

The modelled substance and energy flows were evaluated with a set of environmental indicators, relating to ecological impact categories. Table 65 summarizes the relative comparison of conventional and alternative sanitation systems in these impact categories. The alternative systems offer advantages in nearly all impact categories. The separation of nutrient-rich human excreta from the wastewater treatment disburdens the treatment process and decreases nutrient emissions to surface waters which are responsible for eutrophication.

Table 65: Results of the comparison of ecological impact categories

	Faeces treatment	**Composting**		**Digestion**	
	Greywater treatment	**soil filter**	**SBR**	**soil filter**	**SBR**
Ranking after normalisation	**Impact category**				
1	Eutrophication	++	+	++	++
2	Terrestrial ecotoxicity	++	++	++	++
3	Aquatic ecotoxicity	0	0	+	+
4	Acidification	--	--	-	-
5	Nuclear energy demand	+	-	++	+
6	Depletion of abiotic resources	+	0	++	+
7	Climate change	+	0	++	++
8	Fossil energy demand	+	0	++	+
9	Human toxicity	++	++	++	++

* compared to reference system of conventional wastewater treatment:
++ = better than 20%, + = better than 5%, 0 = no change (+/- 5%), - = worse than 5%, -- = worse than 20%

The utilisation of human excreta for fertilizing purposes lowers the load of toxic heavy metals to agricultural soil in comparison with average mineral fertilizer. Depending on the system configuration, alternative sanitation systems can have a lower energy demand and subsequently cause fewer emissions of climate-active gases. Only the increased emission of acidifying gases (mostly NH_3) represents a considerable drawback compared to the conventional system.

Sensitivity analysis of important system parameters revealed a relative stability of the results towards changes in urine separation efficiency and urine transport distance. The acidification potential of composting systems can be substantially reduced by a complete encapsulating of the process and efficient off-gas cleaning. The phosphate removal in greywater treatment should be as efficient as in the conventional system. Otherwise, a significant increase in the eutrophication potential would change the overall comparison between the systems considerably.

Evaluation of the different processes of alternative sanitation systems

Concerning the treatment of faeces, the digestion process offers significant energy potentials from the partial conversion of organic material into biogas. The high energetic benefit offsets the considerable energy demand of the vacuum system, if biowaste from households is co-digested in the process. The composting of faeces needs less energy for processing, but no energy can be recovered from the faeces. Emissions of nitrogen gases from the composting process can cause an increased acidification potential and global warming. For the agricultural application as fertilizer, both compost and digester residual are comparable in nutrient contents of nitrogen, phosphorus and potassium.

The greywater treatment in a soil filter system needs less energy than a technical process. The natural process is appropriate for the purification of the nutrient-depleted greywater. If the phosphate removal efficiency cannot be maintained on a sufficient level (> 85%), a post-precipitation stage should be implemented. The treatment of faeces filtrate together with the greywater can increase the nutrient loads considerably. In general, the natural process can reach sufficient results in greywater treatment if the required area for a soil filter is available.

The urine separation and application as a fertilizer is a highly useful feature of the alternative systems. The separated urine can be readily applied after an adequate storage time, as it contains a high amount of plant-available nutrients in suitable proportions. A further treatment of urine to reduce transport volume and inactivate potentially harmful pharmaceuticals and other trace organic substances is possible and does not diminish the energetic advantages per se. However, energy-intensive processes like steam stripping and evaporation have to be further optimized to be ecologically reasonable.

Key parameters and sub-processes for future LCA of alternative sanitation systems

In the course of this LCA study, key parameters and sub-processes of system operation could be identified which have a high influence on the decisive ecological impact categories (Table 66). Thus, the data acquisition for future LCA studies of alternative sanitation systems can be simplified and concentrated on important system parts. The listed parameters and sub-processes are not based on a statistical evaluation of their importance, but rather on practical experiences that have been gathered during the work on the substance flow model and the subsequent evaluation.

The expenditures for system construction have only a minor significance for the LCA results. The increased demands of energy and related emissions for construction of multiple-pipe SCST systems are not decisive for the ecological comparison. Hence, the construction phase can be neglected in future simplified LCA studies of sanitation systems.

Table 66: Key parameters and sub-processes for future LCA of alternative sanitation systems

	Vacuum drainage and digestion of faeces	Gravity drainage and composting of faeces	Urine separation	Greywater treatment
Process data	Biogas yield? Amount of biowaste?	Solid-liquid separation Encapsulation/biofilter?	Separation efficiency	Faeces filtrate?
Energy data	Vacuum plant, processing	--	Treatment?	SBR only
Emission data	(NH_3)	NH_3, N_2O	NH_3, N_2O	PO_4 (COD, N)
Transport data	--	To composting	To treatment & farms	--
Construction phase can be neglected in simplified LCA				

Decision support method

Subtask 5 of the SCST project includes the development of a decision support method (DSM) for the choice of an optimal wastewater system under varying boundary conditions. Here, a practicable and well-founded procedure should be set up which allows determining the ecologically most favourable concept. Moreover, the decision support method should be adaptable to specific conditions of other regions.

This milestone could not be fulfilled within the scope of this subproject. During the course of the project, it became clear that the technical realization of alternative sanitation concepts within different boundary conditions cannot be described by simple means (e.g. by a differentiation of “rural”, “sub-urban”, “urban” solutions). It is a rather complex process and is closely linked to the specific situation of the settlement. Consequently, the data which is necessary for a sound ecological assessment via the LCA methodology cannot be easily estimated, but has to be assessed on a case-by-case basis. Without information about the specific settlement, its housing structure, and the surrounding infrastructure, the resulting technical approach for an alternative sanitation concept can only be roughly estimated. Moreover, some features of the investigated alternative systems are not working properly in the pilot plants at WWTP Stahnsdorf (e.g. separation of flush water from faeces, urine transport in pipes (precipitation), separation efficiency of toilets etc). Thus, information about the functionality of these modules and related long-term performance data could not be generalized to a higher level for use in a universal DSM.

The problem of setting up a comprehensive DSM for the choice of an ecologically preferable wastewater treatment system has also been identified by the scientific community. However, the present LCA study seems to be one of the first scientific studies in the field of alternative sanitation with a high level of detail and a precise planning process of the technical implementation within the structure of a real settlement. The close cooperation with the consultant company “Otterwasser” (Mr. Oldenburg) resulted in a realistic assessment of the technical realization and related construction expenditures. Hence, the present LCA model is capable of evaluating the ecological performance of the conventional and several alternative sanitation systems within the specific case study developed during this project. It is not suitable for the deduction of a generalized DSM as claimed by the respective milestone of the project proposal.

In all, the authors of this report think that the present LCA study identified important parameters of sanitation systems for the ecological evaluation. Thus, the development of a more universal DSM in the future should be facilitated with the present work. However, more testing and further development of technical solutions is needed to establish a DSM based on reliable long-term system data. Eventually, it may turn out that the DSM has to be adapted on a case-by-case basis due to the complex influences of the boundary conditions on the technical system realization and performance.

Outlook

In the next steps for an ecological evaluation of alternative sanitation systems, further improvement of the data quality plays an important role. The existing substance flow model can be updated with new data from pilot or full-scale studies to enhance the significance of the presented results. The evaluation method can be developed further with new impact categories (e.g. land use or traffic volume) or more detailed models for existing indicators (e.g. site-specific characterization factors for aquatic and terrestrial eutrophication), and the LCIA results can be verified by statistical methods.

The modular structure of the substance flow model allows an easy implementation of new sub-processes and configurations (e.g. MBR for greywater treatment and subsequent water reuse, low-tech solutions for developing or threshold countries). In all, the presented LCA method is considered to be well-suited for the ecological evaluation of alternative sanitation systems.

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9.1 Composition of water, wastewater and waste fractions

The following tables list the results of an extensive literature research concerning the average composition of human urine and faeces, greywater from kitchen, bath and washing machine, biowaste from households and loppings, and the average composition of drinking water used as flush water in the respective sanitation systems.

Table 67: Volume flow and composition of human urine

Volume flow	unit	1)	2)	3)	4)	5)	6)	7)	8)	9)	SCST-LCA
urine	L/(pe*d)	1,37	1,73		1,57				1,20		**1,5**
Parameters											
dry matter	g/(pe*d)							19,18	72,4		**60**
organic dry matter	g/(pe*d)								39,1		**45**
COD	g/(pe*d)				12,97		6,00	10,19	18,00		**15**
N-total	g/(pe*d)	13,70	8,42		10,80	11,00	10,30	10,49	10,30		**10**
P-total	g/(pe*d)	1,10	0,73	0,8 - 2,0	0,93	1,00		0,68	1,10		**1,00**
K	g/(pe*d)	1,94	2,32	2,70	2,60			2,25	2,20		**2,60**
Na	g/(pe*d)		2,81	3,50			6,00				**3,50**
Ca	g/(pe*d)			0,21					1,40		**0,21**
Mg	g/(pe*d)			0,12							**0,12**
Cl	g/(pe*d)		4,10	4,80			6,50				**4,80**
S-total	g/(pe*d)			1,32				0,63			**0,80**
Cu	mg/(pe*d)		5,80	0,03		0,10		0,05			**0,05**
Zn	mg/(pe*d)		0,46	0,35-0,53		0,04		0,29			**0,25**
Cd	mg/(pe*d)		< = 0,002					0,00		0,0002	**0,0002**
Ni	mg/(pe*d)		0,14	0,00				0,01			**0,04**
Hg	mg/(pe*d)		0,00	0,001-0,009		0,01		0,00		0,0004	**0,0004**
Pb	mg/(pe*d)		< = 0,02	0,04				0,012			**0,01**
Cr	mg/(pe*d)		0,04	0,01		0,01		0,00			**0,01**
AOX	mg/(pe*d)						2,00				**2,00**

1) Lange and Otterpohl, 2000: collection of literature values
2) Jönsson et al., 1997: average values after storage (14d)
3) Average range values from Ciba-Geigy, 1977
4) Fittschen and Hahn, 1998
5) Jönsson and Vinneras, 2003
6) Koppe and Stozek, 1999
7) Calculated from Palmquist and Jönsson, 2003, assumption: 24h attendance of settlement inhabitants
8) Oldenburg, 2002
9) Becker et al., 2002

Table 68: Volume flow and composition of human faeces

Volume flow	unit	1)	2)	3)	5)	6)	7)	8)	SCST-LCA
faeces (wet mass)	kg/(pe*d)	0,14	0,14	0,22	0,13		0,18		**0,14**
Parameters									
dry matter	g/(pe*d)	35,11		50,96	21 - 34		44,70		**45,00**
organic dry matter	g/(pe*d)	29,23			ca. 18		44,80		**42,00**
COD	g/(pe*d)	78,27		4,57		37,00	33,00		**35,00**
BOD	g/(pe*d)			3,35		19,00	11,10		**14,00**
TOC	g/(pe*d)		46,58				21,40		**21,00**
N-total	g/(pe*d)			1,95	1,80	1,90	2,00		**1,50**
P-total	g/(pe*d)	0,96	0,55	0,68	0,31 - 0,77		0,70		**0,50**
K	g/(pe*d)		0,33	0,77	0,44		0,70		**0,55**
Na	g/(pe*d)				0,15				**0,15**
Ca	g/(pe*d)				0,81		1,10		**1,00**
Mg	g/(pe*d)				0,24				**0,20**
Cl	g/(pe*d)				0,06	0,10			**0,06**
S-total	g/(pe*d)			0,21					**0,20**
Cu	mg/(pe*d)			1,74	1,96			1,10	**1,50**
Zn	mg/(pe*d)			46,41	5,1-10,3			10,80	**10,00**
Cd	mg/(pe*d)			0,016	0,16	0,015-0,06		0,01	**0,02**
Ni	mg/(pe*d)			0,22	0,26	0,26		0,07	**0,20**
Hg	mg/(pe*d)			0,01				0,02	**0,02**
Pb	mg/(pe*d)			0,04	0,32			0,02	**0,02**
Cr	mg/(pe*d)			0,13	0,06			0,02	**0,02**
AOX	mg/(pe*d)								**2,00**

1) Calculated from Kujawa-Roeleveld et al., 2003
2) Lange and Otterpohl, 2000: collection of literature values
3) Palmquist and Jönsson, 2003 (faeces + toilet paper)
5) Ciba-Geigy, 1977
6) Koppe and Stozek, 1999: calculation assuming 150 g faeces/(pe*d)
7) Oldenburg, 2002: average values, including toilet paper
8) Vinneras, 2001

Table 69: Volume flow and composition of greywater from households

(*) without tap water loads

Volume flow	unit	1)	2)	3)	4)*	5)	6)	7)	8)	9)	SCST-LCA	SCST-LCA(*)
greywater	L/(pe*d)	110,00				82,19	70,79	69,00			**80,00**	
Parameters												
dry matter	g/(pe*d)	0,04							73,71		**120,00**	**78,400**
COD	g/(pe*d)	47,95	95,00		62,00	35,00	62,76	33,00	56,83	101,53	**60,00**	**60,000**
TOC	g/(pe*d)					15,07			17,40		**18,00**	**17,912**
N-total	g/(pe*d)	1,40	1,30	1,26	0,50	1,00		1,50		1,40	**1,30**	**1,220**
P-total	g/(pe*d)	0,60	0,50	0,30	0,67	0,14		0,15	3,49	0,65	**0,50**	**0,494**
K	g/(pe*d)	0,96	1,85		0,37	2,50					**2,00**	**1,400**
Na	g/(pe*d)		4,00								**6,00**	**3,120**
Ca	g/(pe*d)									14,10	**14,00**	**5,760**
Mg	g/(pe*d)										**3,00**	**2,200**
Cl	g/(pe*d)		10,60							6,95	**7,00**	**5,560**
S-total	g/(pe*d)	1,60	7,70		3,80					7,17	**7,50**	**4,260**
Cu	mg/(pe*d)	6,49		6,00	18,00						**20,00**	**7,200**
Zn	mg/(pe*d)	6,18		33,40	61,00						**46,00**	**16,400**
Cd	mg/(pe*d)	0,02		0,18	0,48					0,08	**0,20**	**0,160**
Ni	mg/(pe*d)	0,66		1,97	6,30						**2,00**	**1,600**
Hg	mg/(pe*d)	0,00		0,02	0,19					0,07	**0,02**	**0,004**
Pb	mg/(pe*d)	0,24		3,00	19,00					18,01	**3,00**	**2,600**
Cr	mg/(pe*d)	0,41		3,01	5,70						**3,00**	**2,600**
AOX	mg/(pe*d)				10,00						**10,00**	**10,000**

1) Palmquist and Jönsson, 2003
2) Koppe and Stozek, 1999: loads from tap water calculated with 200 L/(pe*d)
3) Vinneras, 2001.
4) Schneidmadl, 1999: average values from household wastewater, subtracting urine/faeces/tap water loads
5) Lange and Otterpohl, 2000: collection of literature values
6) Almeida et al., 1999 (back-calculated)
7) Bahlo, 1999
8) Butler et al., 1995: average values from the U.S. 1974-1986
9) Raach et al., 1999 (calculated)

Table 70: Mass flow and composition of organic kitchen waste and loppings

Mass flow	unit	1)	2)	3)	4)	5)	6)	SCST-LCA	unit	1)	7)	8)	SCST-LCA
biowaste	kg/(pe*d)	0,16	0,18			0,18	0,20	**0,20**	kg/(pe*d)	0,33			**0,30**
Parameters for organic kitchen waste									**Parameters for loppings**				
dry matter	g/(pe*d)	50,37					32,00	**50,00**	g/(kg wet mass)	424,62	400,00		**410,00**
organic dry matter	g/(pe*d)	35,81		49,01	30,80	46,09	27,20	**36,00**	g/(kg d.m.)	72,29	70,00		**710,00**
TOC	g/(pe*d)	13,16			10,90		13,60	**13,00**	g/(kg d.m.)	391,22	348,60		**370,00**
N-total	g/(pe*d)	0,82	0,89		1,00		0,43	**0,90**	g/(kg d.m.)	13,93	12,00	6,80	**11,00**
P-total	g/(pe*d)	0,20	0,16	0,17	0,13		0,18	**0,20**	g/(kg d.m.)	9,00	5,28		**5,00**
K	g/(pe*d)	0,60	0,50				0,28	**0,60**	g/(kg d.m.)	12,24	14,94		**13,60**
Na	g/(pe*d)					6,729		**1,20**	g/(kg d.m.)				**0,20**
Ca	g/(pe*d)	0,98					0,35	**1,00**	g/(kg d.m.)	21,95	44,02		**33,00**
Mg	g/(pe*d)	0,22					0,03	**0,22**	g/(kg d.m.)	3,87	4,80		**4,50**
Cl	g/(pe*d)					10,37		**3,00**	g/(kg d.m.)			<0,3	**0,30**
S-total	g/(pe*d)		0,09					**0,10**	g/(kg d.m.)			0,50	**0,50**
Cu	mg/(pe*d)	1,06	0,44				0,34	**1,00**	mg/(kg d.m.)	27,82	10,00	10,00	**19,00**
Zn	mg/(pe*d)	7,28	1,85				0,98	**7,30**	mg/(kg d.m.)	152,82	60,00	57,00	**110,00**
Cd	mg/(pe*d)	0,01	0,007				0,00	**0,01**	mg/(kg d.m.)	0,60	0,27	< 2,2	**0,40**
Ni	mg/(pe*d)		0,24				0,30	**0,20**	mg/(kg d.m.)		3,70		**3,70**
Hg	mg/(pe*d)	0,01	0,01				0,00	**0,01**	mg/(kg d.m.)	0,15	0,23		**0,20**
Pb	mg/(pe*d)	1,06	0,11				0,06	**0,60**	mg/(kg d.m.)	47,27	4,80	< 6,6	**4,80**
Cr	mg/(pe*d)		0,50				0,05	**0,50**	mg/(kg d.m.)		4,60	54,00	**4,60**

1) Average value, calculated from Wintzer et al., 1996
2) Palmquist and Jönsson, 2003
3) Expected values in Kujawa-Roeleveld et al., 2003
4) Paik et al. (1999), cited in Kujawa-Roeleveld et al., 2003: assumption 0,2 L/(pe*d)
5) Kübler et al. (1999), cited in Kujawa-Roeleveld et al., 2003: assumption 0,18 L/(pe*d)
6) Vogt et al., 2002: calculated assuming 0,2 kg/(pe*d)
7) Vogt et al., 2002
8) Wolff, 2004

9.2 *Characterization factors for Life Cycle Impact Assessment*

Table 71: LCIA characterization factors for airborne emissions: GWP, AP, EP

NAME	**MED**	**GWP100**	**AP**	**AP_reg_thr**	**EP**
UNIT		kg CO2-eq/kg	kg SO2-eq/kg	kg SO2-eq/kg	kg PO4-eq/kg
DESCRIPTION		Global Warming Potential (100a) [1]	Acidification Potential [2]	Acidification Potential for Germany incl treshold concentration [3]	Eutrophication Potential [2,4] (*)
CH4	air	2,30E+01			
Perfluoroethane	air	1,19E+04			
Perfluoromethane	air	5,70E+03			
NH3	air		1,88E+00	7,70E-01	(3,50E-01)
HCl	air		8,80E-01	8,80E-01	
N2O	air	2,96E+02			
HF	air		1,60E+00	1,60E+00	
CO2, fossil	air	1,00E+00			
CO	air	1,58E+00			
NOx	air		7,00E-01	1,90E-01	(1,30E-01)
SO2	air		1,00E+00	6,00E-01	
H2S	air		1,88E+00	1,88E+00	

[1]: Houghton et al., 1996
[2]: Heijungs et al., 1992
[3]: Huijbregts, 1999
[4]: Guinée et al., 2002

(*): for eutrophication potential, both cases (with and without nitrogen emissions) are discussed (see chapter 2.2.1 for more information)

Table 72: LCIA characterization factors for airborne emissions: HTP, FAETP, TETP, AET, TET

NAME	MED	HTP	FAETP	TETP	AET	TET
UNIT		kg 1,4-dichlorobenzene-eq/kg			kg of triethylene-glycol-eq/kg	
DESCRIPTION		Human Toxicity Potential [1]	Freshwater Aquatic Ecotoxicity Potential [1]	Terrestrial Ecotoxicity Potential [1]	Aquatic Ecotoxicity [2]	Terrestrial Ecotoxicity [2]
Particles	air	8,20E-01				
Dust (PM10)	air	8,20E-01				
Sb	air	6,71E+03	3,72E+00	6,11E-01	2,01E+04	2,96E+05
As	air	3,48E+05	4,95E+01	1,61E+03	4,19E+05	5,48E+04
Be	air	2,27E+05	1,71E+04	1,77E+03		
Pb	air	4,67E+02	2,40E+00	1,57E+01	1,31E+05	4,01E+04
Cd	air	1,45E+05	2,89E+02	8,12E+01	9,12E+05	4,28E+05
Cr III	air	6,47E+02	1,92E+00	3,03E+03		
Cr VI	air	3,43E+06	7,69E+00	3,03E+03		
Cr (III+VI)	air	3,49E+04	1,98E+00	3,03E+03	3,82E+05	6,70E+04
Co	air	1,75E+04	6,39E+02	1,09E+02		
Cu	air	4,30E+03	2,22E+02	6,99E+00	1,18E+06	2,94E+06
Ni	air	3,50E+04	6,29E+02	1,16E+02	5,63E+05	1,79E+05
Hg	air	6,01E+03	3,17E+02	2,83E+04	3,84E+06	7,86E+05
Se	air	4,77E+04	5,46E+02	5,35E+01	9,32E+03	4,78E+05
TI	air	4,32E+05	1,55E+03	3,40E+02		
V	air	6,24E+03	1,73E+03	6,65E+02		
Zn	air	1,04E+02	1,78E+01	1,20E+01	1,01E+06	2,04E+05
Sn	air	1,73E+00	2,54E+00	1,44E+01		
Benzo(a)pyrene	air	5,72E+05	8,78E+01	2,41E-01	2,42E+01	2,50E+04
Benzene	air	1,90E+03	8,37E-05	1,56E-05	4,72E-03	2,44E-02
Chlorobenzene	air	9,23E+00	4,68E-04	7,29E-04	3,36E-01	5,07E+00
Formaldehyde	air	8,31E-01	8,26E+00	9,40E-01	2,54E+00	1,67E+00
PAH w/o B(a)P	air	5,72E+05	1,72E+02	1,02E+00		
PAH, unspec.	air	5,72E+05	1,72E+02	1,02E+00		
PCDD, PCDF	air	1,93E+09	2,13E+06	1,20E+04	2,59E+02	3,94E+05
NH3	air	1,00E-01			9,80E+00	3,90E+00
HCl	air	5,00E-01				
HF	air	2,85E+03	4,64E+00	2,95E-03	7,84E-01	1,79E+02
NOx	air	1,20E+00				
SO2	air	9,60E-02				
H2S	air	2,20E-01				

[1]: Huijbregts et al., 2000
[2]: Jolliet et al., 2003

Table 73: LCIA characterization factors for emissions in soil: HTP, FAETP, TETP, AET, TET

NAME	**MED**	**HTP**	**FAETP**	**TETP**	**TET**	**AET**
UNIT		kg 1,4-dichlorobenzene-eq/kg			kg of triethylene-glycol-eq/kg	
DESCRIPTION		Human Toxicity Potential [1]	Freshwater Aquatic Ecotoxicity Potential [1]	Terrestrial Ecotoxicity Potential [1]	Terrestrial Ecotoxicity [2]	Aquatic Ecotoxicity [2]
As	soil	3,18E+04	1,34E+02	3,34E+03	2,43E+06	3,87E+05
Pb	soil	3,28E+03	6,53E+00	2,34E+01	7,54E+05	2,64E+05
Cd	soil	1,96E+04	7,76E+02	1,67E+02	5,28E+06	2,91E+06
Cr III	soil	5,13E+03	5,25E+00	6,30E+03		
Cr VI	soil	8,55E+03	2,10E+01	6,30E+03		
Cr (III+VI)	soil	5,16E+03	5,41E+00	6,30E+03	2,25E+06	4,49E+05
Co	soil	2,39E+03	1,71E+03	2,23E+02		
Cu	soil	9,39E+01	5,95E+02	1,44E+01	6,92E+06	2,04E+07
Ni	soil	2,68E+03	1,69E+03	2,39E+02	3,30E+06	1,26E+06
Hg	soil	5,92E+03	8,48E+02	5,60E+04	2,65E+07	1,58E+07
Zn	soil	6,37E+01	4,77E+01	2,46E+01	5,91E+06	1,40E+06
Sn	soil	1,31E+01	6,90E+00	2,98E+01		

[1]: Huijbregts et al., 2000
[2]: Jolliet et al., 2003

Table 74: LCIA characterization factors for emissions in water: HTP, FAETP, TETP, AET, TET

NAME	**MED**	**HTP**	**FAETP**	**TETP**	**TET**	**AET**
UNIT		kg 1,4-dichlorobenzene-eq/kg			kg of triethylene-glycol-eq/kg	
DESCRIPTION		Human Toxicity Potential [1]	Freshwater Aquatic Ecotoxicity Potential [1]	Terrestrial Ecotoxicity Potential [1]	Terrestrial Ecotoxicity [2]	Aquatic Ecotoxicity [2]
Sb	water	5,14E+03	1,97E+01	1,66E-20	1,22E-09	2,10E+06
As	water	9,51E+02	2,07E+02	1,04E-17		3,88E+05
Ba	water	6,30E+02	2,28E+02	5,08E-19	1,53E-10	8,05E+04
Be	water	1,40E+04	9,13E+04	3,30E-16		
Pb	water	1,23E+01	9,62E+00	4,77E-22		2,64E+05
Cd	water	2,29E+01	1,52E+03	1,42E-20	1,52E-09	2,92E+06
Cr III	water	2,05E+00	6,91E+00	2,27E-19		
Cr VI	water	3,42E+00	2,77E+01	2,27E-19		
Cr (III+VI)	water	2,07E+00	7,12E+00	2,27E-19		4,53E+05
Co	water	9,67E+01	3,41E+03	2,69E-18		
Cu	water	1,34E+00	1,16E+03	4,06E-21		2,06E+07
Mo	water	5,51E+03	4,76E+02	2,31E-18		
Ni	water	3,31E+02	3,24E+03	1,03E-18		1,27E+06
Hg	water	1,43E+03	1,72E+03	9,30E+02	6,93E-08	1,58E+07
Se	water	5,60E+04	2,92E+03	1,55E-17		3,40E+06
V	water	3,16E+03	8,97E+03	1,02E-17		
Zn	water	5,84E-01	9,17E+01	2,53E-21		1,40E+06
Sn	water	1,73E-02	1,02E+01	7,86E-22		
Fluoride	water	3,42E+03	1,81E+01	4,28E-05		
Fluorine	water	3,42E+03	1,81E+01	4,28E-05		
Ammonium	water				2,18E-02	4,98E+02
NH3-N	water				2,80E-02	6,40E+02
NH4-N	water				2,80E-02	6,40E+02
Benzo(a)pyrene	water	2,80E+05	2,52E+05	2,53E-03	2,27E-01	1,71E+06
PAH w/o B(a)P	water	2,80E+05	2,75E+04	2,12E-03		
Phenols	water	4,92E-02	2,37E+02	2,49E-06		

[1]: Huijbregts et al., 2000
[2]: Jolliet et al., 2003

Table 75: LCIA characterization factors for emissions in water: EP, EP-limP

NAME	**MED**	EP	EP-limP
UNIT		kg PO4-eq/kg	kg PO4-eq/kg
DESCRIPTION		Eutrophication Potential [1]	Aquatic Eutrophication [2]
BOD5	water	4,40E-02	4,40E-02
COD	water	2,20E-02	2,20E-02
TOC	water	4,40E-02	4,40E-02
NH3	water	3,50E-01	
NH4	water	3,30E-01	
NH3-N	water	4,20E-01	
NH4-N	water	4,20E-01	
NO3	water	1,00E-01	
NO3-N	water	4,43E-01	
N species as N	water	4,20E-01	
HNO3	water	1,00E-01	
PO4	water	1,00E+00	1,00E+00
PO4-P	water	3,06E+00	3,06E+00
Phosphates (as P2O5)	water	1,34E+00	1,34E+00
Phosphorus	water	3,06E+00	3,06E+00
P species as P	water	3,06E+00	3,06E+00

[1] : Guinée et al., 2002
[2] : Jolliet et al., 2003

Table 76: LCIA characterization factors for extraction of resources: ADP

NAME	**MED**	ADP
UNIT		kg antimony-eq/kg
DESCRIPTION		Abiotic Depletion Potential [1]
lignite	resource	6,71E-03
Natural gas	resource	1,87E-02
Crude oil	resource	2,01E-02
Coal	resource	1,34E-02
Hard coal	resource	1,34E-02
Uranium	resource	2,87E-03
Lead	resource	6,77E-04
Iron	resource	4,80E-08
Ferromanganese	resource	6,20E-06
Zinc	resource	3,95E-05
Bauxite	resource	2,10E-09
Sulphur	resource	3,58E-04
Raw phosphate	resource	9,29E-06
Nickel ore, sulphured	resource	5,38E-06
Nickel ore, lateritic	resource	1,08E-06
Raw potassium	resource	3,76E-09
Copper ore (0,99% Cu)	resource	2,20E-05
Chromium ore	resource	2,58E-04
Chromium	resource	8,58E-04

[1]: Guinée et al., 2002

9.3 Normalisation data

Normalisation data is provided within the CML method for the following geographical and temporal scope: Netherlands 1997, Western Europe 1995, and the World 1995. However, this LCA study is related to conditions in Germany today or in the near future. It was decided to generate normalisation data based on the relevant impact categories and on the latest emission data for Germany. In the following, the generation of normalisation data is described in detail for each indicator value.

Basic definitions for normalisation

Geographical scope: Germany
Population: 82,532 mio
Time: reference year 2002 (if available, otherwise the latest possible data)

Cumulated energy demand

Table 77: Normalisation data for cumulated energy demand

Type	Annual consumption in Germany 2004 (BMWA, 2005)	CED per inhabitant
	[PJ/a]	[GJ/(pe*a)]
CED fossil	12,081	146
CED nuclear	1,823	22
CED renewable	164	2
CED misc	370	5
CED total	14,438	175

Climate change

Airborne emissions are responsible for climate change. Some substances are neglected (i.e. CFC etc), because they are not included in the inventory analysis of this study. IPCC characterization factors are taken from Guinée et al., 2002 and are related to a time horizon of 100 years (GWP_{100a}). Emission data is taken from UBA, 2003.

Table 78: Normalisation data for climate change

Substance	Annual emissions in Germany 2002		Characterization factor	GWP per inhabitant
	[1000 Mg/a]	[kg/(pe*a)]	[kg CO2-eq/kg]	[kg CO2-eq/a]
CO2	864,117	10,470	1	10470
CO	4,318	52	1.58	82
CH4	3,878	47	23	1081
N2O	180	2.2	296	651
total				12,279

Acidification

Airborne emissions are responsible for acidification. Some substances are neglected (i.e. HCl) due to lack of emission data. Emission data is taken from UBA, 2003, except for HF where it is estimated from emissions in 1991 (124,000 t/a). Regional characterization factors for Germany considering a threshold effect are applied (Huijbregts, 1999).

Table 79: Normalisation data for acidification

Substance	Annual emissions in Germany 2002		Characterization factor	AP per inhabitant
	[1000 Mg/a]	[kg/(pe*a)]	[kg SO2-eq/kg]	[kg SO2-eq/a]
NH3	607,000	7.4	0.77	5.7
HF	86,800	1.1	1.1	1.2
SO2	608,000	7.4	0.6	4.4
NOx (as NO2)	1479,000	17.9	0.19	3.4
total				14,6

Eutrophication

Normalisation data is generated for eutrophication potential with (EP) and without (EP-limP) accounting for nitrogen emissions. Emission data is taken from UBA, 2003. For COD, emission data is estimated with 120 g COD/(pe*d) (ATV, 2000) and an average elimination of 90% in wastewater plants.

Table 80: Normalisation data for eutrophication

Substance	Annual emissions in Germany 2002		Characterization factor		EP per inhabitant	
	[1000 Mg/a]	[kg/(pe*a)]	[kg PO4-eq/kg]		[kg SO2-eq/a]	
			EP	EP-limP	EP	EP-limP
Air						
NH3	607,000	7.4	0.35		2.6	
NOx	1479,000	17.9	0.13		2.3	
Water						
COD	360,500	4.4	0.022	0.022	0.1	0.1
P species as P	33,164	0.4	3.06	3.06	1.2	1.2
N species as N	687,960	8.3	0.42		3.5	
total					9.7	1.3

Human and ecotoxicity

Table 81: Normalisation data for human and ecotoxicity

Substance	**Annual emission in Germany**		**Source**	**Indicator per inhabitant**				
	[Mg/a]	[g/(pe*a]		**HTP**	**FAETP**	**TETP**	**AET**	**TET**
Air				[kg 1,4-DCB-eq/a]			[kg TEG-eq/a]	
As	33	0.40	1)	1.15E+10	1.63E+06	5.31E+07	1.81E+09	1.38E+10
Be	2	0.02	1)	4.53E+08	3.43E+07	3.53E+06	0.00E+00	0.00E+00
Pb	632	7.66	1)	2.95E+08	1.52E+06	9.90E+06	2.53E+10	8.28E+10
Cd	11	0.13	1)	1.60E+09	3.18E+06	8.94E+05	4.71E+09	1.00E+10
Cr (III)	113.85	1.38	2)	7.36E+07	2.19E+05	3.45E+08	0.00E+00	0.00E+00
Cr (VI)	1,15	0.01	2)	3.94E+09	8.84E+03	3.49E+06	0.00E+00	0.00E+00
Cr (III+VI)	115	1.39	1)	4.01E+09	2.28E+05	3.49E+08	6.98E+09	4.39E+10
Co	12	0.15	1)	2.10E+08	7.67E+06	1.30E+06	6.49E+09	2.54E+09
Cu	79	0.96	1)	3.39E+08	1.75E+07	5.52E+05	2.32E+11	9.32E+10
Ni	159	1.93	1)	5.57E+09	1.00E+08	1.85E+07	2.85E+10	8.95E+10
Hg	31	0.38	1)	1.86E+08	9.82E+06	8.78E+08	2.44E+10	1.19E+11
Se	25	0.30	1)	1.19E+09	1.37E+07	1.34E+06	1.20E+10	2.33E+08
Tl	8	0.10	1)	3.46E+09	1.24E+07	2.72E+06	0.00E+00	0.00E+00
Zn	452	5.48	1)	4.72E+07	8.04E+06	5.41E+06	9.22E+10	4.57E+11
Benzo(a)pyrene	13.757	0.17	3)	7.87E+09	1.21E+06	3.32E+03	3.44E+08	3.33E+05
Benzene	42900	519.80	3)	8.15E+10	3.59E+03	6.68E+02	1.05E+06	2.02E+05
PAH w/o B(a)P	382.243	4.63	1)	2.19E+11	6.57E+07	3.90E+05	0.00E+00	0.00E+00
PCDD, PCDF	0.00031	4.00E-6	1)	5.98E+08	6.57E+05	3.71E+03	1.22E+05	8.00E+01
NH3	607000	7,354	4)	6.07E+07	0.00E+00	0.00E+00	0.00E+00	0.00E+00
HF	86800	1051	5)	2.47E+11	4.03E+08	2.56E+05	0.00E+00	0.00E+00
SO2	608000	7367	4)	5.84E+07	0.00E+00	0.00E+00	0.00E+00	0.00E+00
NOx (as NO2)	1479000	17920	4)	1.77E+09	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Dust (PM10)	247000	2993	4)	2.03E+08	0.00E+00	0.00E+00	0.00E+00	0.00E+00

(1): UBA, 2003 (data from 1995) (2): assumed: 99% Cr^{III} and 1% Cr^{VI} (3): UBA, 2000 (data from 1993/94)

(4): UBA, 2003 (5): UBA, 2003 assumed from data of 1991

Table 82: Normalisation data for human and ecotoxicity (ctd)

Substance	Annual emission in Germany		Source	Indicator per inhabitant				
	[Mg/a]	[g/(pe*a)]		HTP	FAETP	TETP	AET	TET
Water					[kg 1,4-DCB-eq/a]		[kg TEG-eq/a]	
Pb	287.4	3.48	1)	1.15E+10	1.63E+06	5.31E+07	1.81E+09	1.38E+10
Cd	9.9	0.12	1)	4.53E+08	3.43E+07	3.53E+06	0.00E+00	0.00E+00
Cr (III+VI)	261.2	3.17	1)	7.36E+07	2.19E+05	3.45E+08	0.00E+00	0.00E+00
Cu	594.6	7.21	1)	3.94E+09	8.84E+03	3.49E+06	0.00E+00	0.00E+00
Ni	324	3.93	1)	4.01E+09	2.28E+05	3.49E+08	6.98E+09	4.39E+10
Hg	4.2	0.05	1)	2.10E+08	7.67E+06	1.30E+06	6.49E+09	2.54E+09
Zn	2931	35.51	1)	3.39E+08	1.75E+07	5.52E+05	2.32E+11	9.32E+10
Soil								
Pb	1464	17.74	2)	1.86E+08	9.82E+06	8.78E+08	2.44E+10	1.19E+11
Cd	84	1.02	2)	1.19E+09	1.37E+07	1.34E+06	1.20E+10	2.33E+08
Cr (III+VI)	561	6.80	3)	3.46E+09	1.24E+07	2.72E+06	0.00E+00	0.00E+00
Cu	4577	55.46	2)	4.72E+07	8.04E+06	5.41E+06	9.22E+10	4.57E+11
Ni	476	5.77	3)	7.87E+09	1.21E+06	3.32E+03	3.44E+08	3.33E+05
Hg	3.06	0.04	4)	8.15E+10	3.59E+03	6.68E+02	1.05E+06	2.02E+05
Zn	21237	257.32	2)	2.19E+11	6.57E+07	3.90E+05	0.00E+00	0.00E+00
TOTAL (Air + Water + Soil)				**7266**	**88.9**	**70.1**	**1720600**	**1969620**

(1): Fuchs et al., 2002 (2): Wilcke and Döhler, 1995(3): calculated from Eckel et al., 2005
(4): estimation according to data from Bannick et al., 2001

Generating normalisation data for human and ecotoxicity is a difficult task. Emission data for various substances in various media (air, water, soil) have to be collected. Even though the data symmetry (time, quality) is fairly poor, the generated data has the advantage to account only those substances which are included in the inventory analysis. Throughout the normalisation, it is clearly visible that the different underlying toxicity models for the set of indicators (HTP, FAETP, TETP from CML method and AET, TET from IMPACT2002+) result in a totally different contribution of each substance to the specific indicators.

9.4 Data for construction of sanitary systems

9.4.1 Map of settlement area

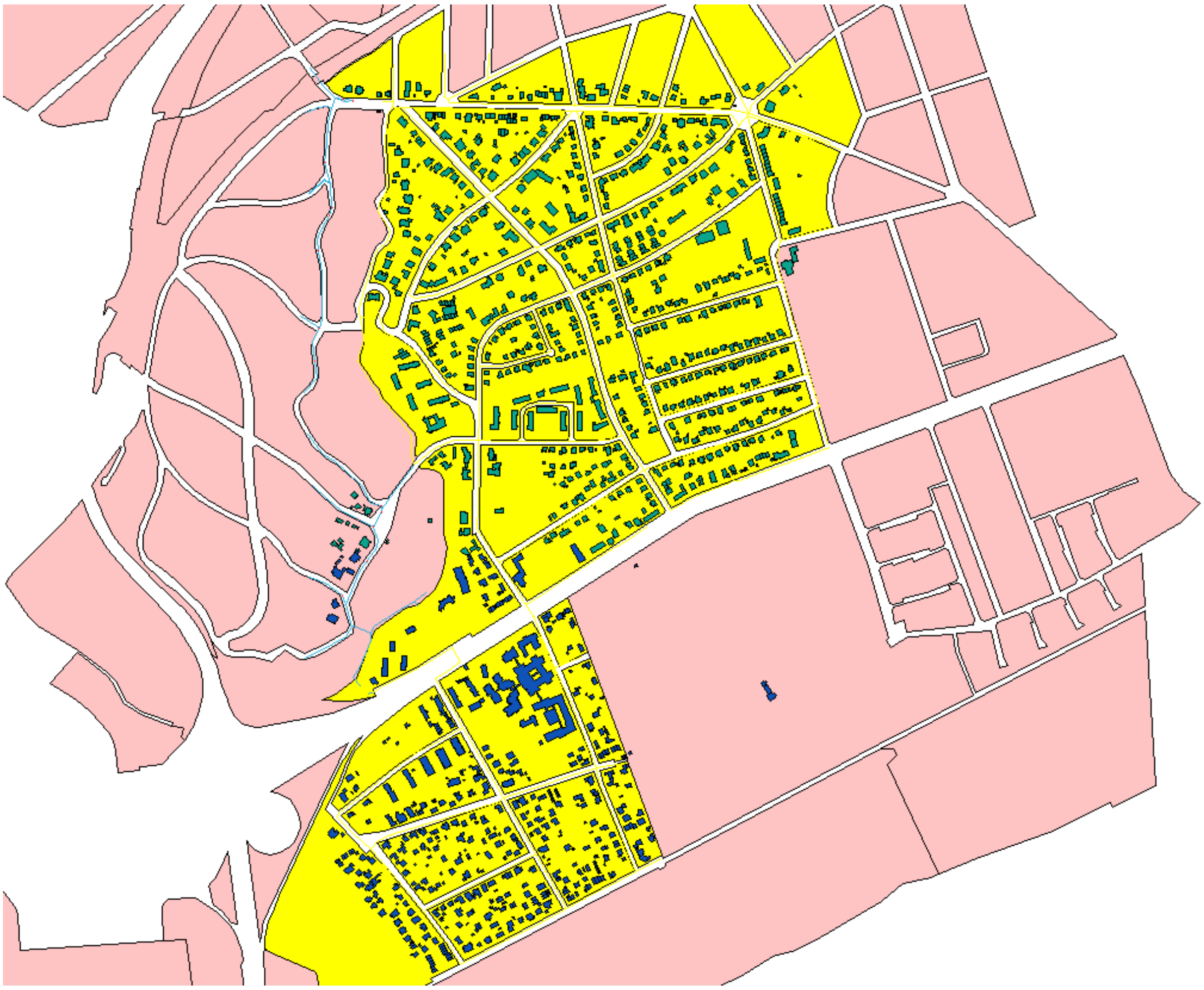

Figure 58: Map of the settlement (Berlin-Nicolassee)

Source: GIS data from Berliner Wasserbetriebe, screenshot from ARCVIEW® software

The study area is marked in yellow.

9.4.2 Sanitary installations

The sanitary piping of prototype houses and apartments is done according to the German norm for dimensioning of sanitary installations (DIN EN 12056-2). Figure 59 shows the prototype house and Figure 60 the prototype apartment with the conventional sanitary system. Details of the calculation of sanitary in-house-piping for conventional and alternative systems are presented in Table 83 and Table 84.

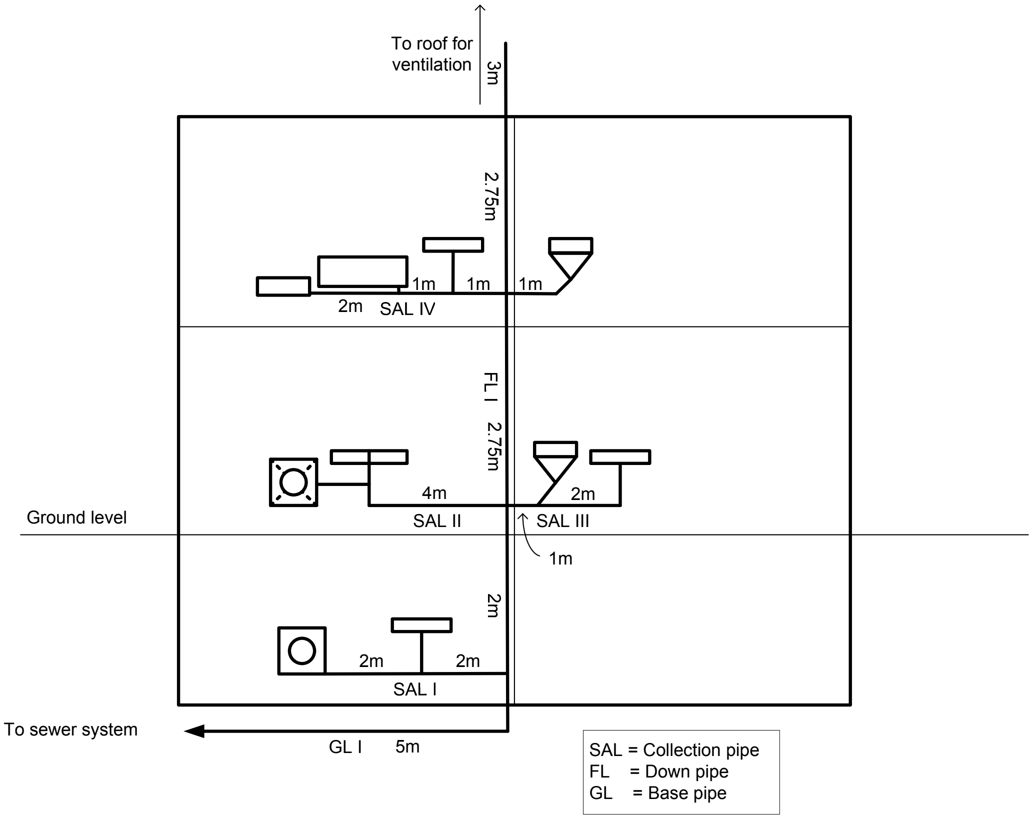

Figure 59: Prototype house for dimensioning of sanitary installations

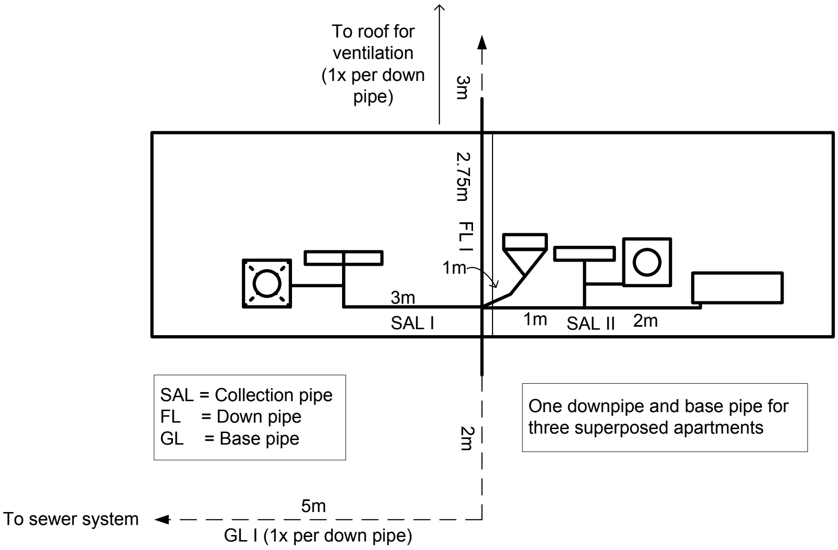

Figure 60: Prototype apartment for dimensioning of sanitary installations

Table 83: Calculation of sanitary installations (conventional system)

Building	Line	Pipe material and diameter	Length [m]	Sum [m]	Total length [m]
House	SAL I	PP 50	4	14	12180
	SAL II	PP 50	4		
	SAL III	PP 50	2		
	SAL IV	PP 50	4		
	SAL III	PP 100	1	12.5	10875
	SAL IV	PP 100	1		
	Down pipe	PP 100	10.5		
	Base pipe	PP 150	5	5	4350
Apartment	SAL I	Cast iron 50	3	6	7020
	SAL II	Cast iron 50	3		
	SAL II	Cast iron 100	1	16.25	6337.5
	Down pipe (*)	Cast iron 100	13.25		
	Base pipe (*)	PP 150	5	15	1950

(*) one down pipe and base pipe for three superposed apartments

Table 84: Calculation of sanitary installations (alternative systems)

Building	System**	Line	Pipe material and diameter	Length [m]	Sum [m]	Total length [m]
House	BW	SAL III	PP 100	1	12.5	10875
		SAL IV	PP 100	1		
		Down pipe	PP 100	10.5		
		Base pipe	PP 150	5	5	4350
	GW	SAL I	PP 50	4	15	13050
		SAL II	PP 50	4		
		SAL III	PP 50	3		
		SAL IV	PP 50	4		
		Down pipe	PP 70	6.25	6.25	5437.5
		Base pipe	PP 100	5	5	4350
	US	SAL III	PP 50	1	2	1740
		SAL IV	PP 50	1		
		Down pipe	PP 70	6.25	6.25	5437,5
		Base pipe	PP 100	5	5	4350
	VC	Total	PE 50	12	12	10440
Apartment	BW	SAL II	Cast iron 100	1	16.25	6337,5
		Down pipe*	Cast iron 100	13.25		
		Base pipe*	Cast iron 150	5	15	1950
	GW	SAL I	Cast iron 50	3	6	7020
		SAL II	Cast iron 50	3		
		Down pipe*	Cast iron 70	9	27	3510
		Base pipe*	PP 100	5	15	1950
	US	SAL II	PP 50	1	1	1170
		Down pipe*	PP 70	9	27	3510
		Base pipe*	PP 100	5	15	1950
	VC	Total	PE 50	17	17	6630

(*) one down pipe and base pipe for three superposed apartments

(**) US: urine separation, BW: brown water, GW: greywater, VC: vacuum

9.4.3 Drainage systems

The layout of the drainage systems is designed following common German regulations. Figure 61 shows an exemplary map of the conventional sewer system in the northern part of the study area. Table 85 lists the piping length and material, Table 86 summarizes the additional components for the different scenarios. Table 87 shows the weight of single components together with the data sources. Material weight data is estimated from manufacturer product information (pipes: www.geberit.de, www.simona.de, www.euroceramic.de, www.dueker.de, www.ostendorf-kunststoffe.com; shafts: www.mallbeton.de)

Figure 61: Exemplary layout of the conventional sewer system in the northern part of the study area (Red: sewage pipes, green: sections of drainage area)

Table 85: Pipe dimensions and materials for drainage system

Material	Ø	Weight	REF	COMP	VAC	Weight factor*
	[mm]	[kg/m]	[m]	[m]	[m]	[%]
House connections						
Vitrified clay	150	24	10000	10000	0	10
Polypropylene	150	1.943	0	10000	10000	15
Polyethylene	50	0.457	0	0	10000	20
	150	2.46	0	10000	10000	20
Drainage						
Vitrified clay	150	24	11250	11108	0	10
	200	36	1010	0	0	10
	250	51	645	0	0	10
	300	67	370	0	0	10
	400	104	530	0	0	10
Polyethylene	50	0.457	0	14866	7050	15
	65	0.727	0	0	14185	15
	100	2.18	0	0	1364	15
	150	2.46	0	13168	13168	15
	200	4.82	0	470	470	15
Polypropylene	150	1.943	0	11108	11108	20

(*) proportional addition for bends, tees etc (PP: including 5% for plug-in connections)

Table 86: Additional components of drainage systems

System	Component	Material	Dimensions		Quantity
				Depth [m]	
Conventional	House shafts	Concrete	DN 1000	2	1000
	Inspection chambers	Concrete	DN 1000	3	342
Brownwater	House shafts	Concrete	DN 1000	2	-*
	Inspection chambers	Concrete	DN 1000	3	-*
	Pumping shafts	Concrete	DN 1000	3	32
Greywater	House shafts	Concrete	DN 1000	2	1000
	Inspection chambers	Concrete	DN 1000	3	273
Urine	Inspection chambers	Plastic	DN 400	3	223
	Pumping shafts	Concrete	DN 1000	3	32
	Holding tanks	GRP	12 m³	-	12
Vacuum	Inspection chambers	Concrete	DN 1000	1	67

(*) together with greywater in one house shaft and inspection chamber

Table 87: Material weight of shafts

Component	Material	Dimensions		Weight
			Depth [m]	[kg]
Inspection chambers	Concrete	DN 1000	3	3050
	Cast iron			35
House shafts	Concrete	DN 1000	2	1610
	Cast iron			26
Pumping shafts	Concrete	DN 1000	3	3050
	Cast iron			35
Urine inspection shafts	Polypropylene	DN 400	3	40
Manhole covers	Concrete	DN 800		80/75*
	Cast iron			40/120*
(*) class B125/D400: house shafts are B125, rest is D400				

9.4.4 Treatment facilities

The following section lists the material data for the different treatment facilities. The process design is heavily depending on local conditions and specific process layout. Therefore, the assessment of the material demand for the treatment facilities can only be a rough estimation of the actual material demand. However, the contribution of the facilities to the overall system construction expenditures may be estimated with the following data. The difficult acquisition of proper construction data for the various processing units results in certain qualified estimates to be made for the material demand. Detailed construction data is not available for all system parts, so that manufacturer product information has to be used as a basis for estimations.

Conventional wastewater treatment plant/SBR for greywater treatment

Construction data for the conventional wastewater treatment plant is taken from a detailed material inventory of a German activated sludge plant (design dimension: 21000 inhabitant equivalents) (Schneidmadl, 1999). This data is converted to the present plant dimension (~ 5000 inhab. equiv.) by recalculating the material demand in relation to the volume of the treated influent. For the technical greywater plant (SBR process), adequate detailed material data could not be acquired. Hence, it is assumed that both processes (conventional WWTP and greywater treatment) have a comparable material demand in relation to the treated volume, and the greywater treatment plant is described with the same data set.

Table 88: Construction data for conventional wastewater treatment plant and SBR plants for greywater treatment

	Marienheide (Schneidmadl, 1999)		**SCST**		
			Ref	Comp_Tech	Vac_Tech
Inhabitant equivalents	21000		4891	4891	4891
Influent volume [m^3/d]	7540		543.4^1	543.4^2	430.4^3
Material	**kg**	**kg/(m^3*a)**	**kg**	**kg**	**kg**
Aluminium	3090	0.00004	238	238	189
Concrete	30809720	0.37317	2220447	2220447	1758738
Glas	1130	0.00001	60	60	47
Limestone	602580	0.00730	43437	43437	34405
Copper	13310	0.00016	952	952	754
Polyethylene HD	28820	0.00035	2083	2083	1650
Steel, high-alloy	57390	0.00070	4165	4165	3299
Steel, non-alloy	498020	0.00603	35880	35880	28419
Excavation	96382600	1.16738	6946179	6946179	5501824

Service life for all variants: 30 a

1) 101 L/(pe*d), plus 10% as excess allowance
2) greywater including faeces filtrate, plus 10% excess allowance
3) greywater, plus 10% excess allowance

Soil filter for greywater treatment

Greywater and faeces filtrate are treated in a planted soil filter in scenarios Comp_Nat and Vac_Nat. For the whole settlement, three soil filters are designed according to the system dimensioning by Otterwasser.

Each soil filter consists of a sedimentation tank (HRT~ 3h), a feeding pump in a pumping sump, and a planted soil filter with ground sealing (PE foil + PES fleece) and feeding and drainage pipes. For the dimensioning, the necessary surface area for the soil filter is set according to recommendations of ATV (ATV, 1998) and operational experience from Lübeck-Flintenbreite (Oldenburg, 2002) and Stahnsdorf. In scenario Vac_Nat, the soil filter treats only greywater (80 L/(pe*d)), and the required surface area is set to 2 m^2/inhabitant. In case of combined treatment of greywater and faeces filtrate (Comp_Nat), the necessary surface area is increased to 2.5 m^2/ inhabitant to prevent hydraulic overloading and clogging of the filter.

Whereas the material demand for the sedimentation tank is estimated from manufacturer data, the soil filter components are adopted from existing pilot plants or are qualified estimates from experts.

Table 89: Material data for system components of a soil filter

Component	Material	Value	Unit	Source
Sedimentation tank	Concrete	29582	kg	Mall, 2005
(39 m³)	Steel, non-alloy	444	kg	
	Excavation	117	m³	Estimation
Pumps (2x)	Cast iron	100	kg	Bengtsson et al., 1997
Pumping sump	Polyethylene	110	kg	Romold, 2004
	Cast iron	100	kg	
Soil filter	Excavation	1	m³/m²	Estimation
	Sand	1	m³/m²	see operation
	PE foil	1.4	kg/m²	Spiess-Wallbaum, 2002
	PES fleece	0.16	kg/m²	
Pipe DN50 (Feed)	PE	0.5	kg/m²	Oldenburg, 2002 pipes: Simona, 2005
Pipe DN100 (Drainage)	PE	1.44	kg/m²	

Urine tanks

Urine is interim stored in underground plastic tanks (material: glass fibre reinforced plastic). The required volume of the tanks is calculated from the daily amount of urine and a minimum holding time of 14 days. The urine is withdrawn from the tanks by a suction vehicle and transported to the nearby farms for storage (> 0.5 a) and subsequent application on the fields.

Table 90: Material data of interim storage tanks for separated urine

Components	Material	Weight [kg]	Quantity
Interim storage tank (12 m³)	Glass fibre reinforced plastic	800	9
Interim storage tank (11 m³)		730	3
Source: Mannschott, 2005			

Solid-liquid separation system

For the separation of flush water from faeces in composting scenarios, three technical solid-liquid separators are operated within the settlement. At present, a reliable and efficient system is not commercially available. Thus, the material demand for such a separator has to be roughly estimated.

It is assumed that an intermediate storage tank (10 m³) is provided for compensation of volume variations. The separator and subsequent thickener are protected by an automated rake system against damage. The filtrate is pumped in a pressure line (50 m) to the adjacent soil filter or SBR plant. For redundancy, all pumps are built twice. The separated solid matter is stored in dewatering containers before it is brought to the composting plant by truck.

Table 91: Material data for a solid-liquid separator

Component	Material	Value	Unit	Quantity	Source
Pumps	Cast iron	100	kg	6	Bengtsson et al., 1997
Storage tank (10 m³)	Concrete	7500	kg	1	Mall, 2005
	Steel, non-alloy	115	kg		
Flocculation aid dosage	Polyethylene	100	kg	1	Estimation
Rake system	Steel, high alloy	250	kg	1	Estimation
Separator/thickener	Steel, high alloy	500	kg	1	Estimation
Dewatering container	Steel, non-alloy	1000	kg	5	Estimation
Pressure pipe DN50	PE	0.457	kg/m	100 m	Simona, 2005

Vacuum system

The vacuum system for the transport of faeces comprises of three vacuum stations which are distributed across the settlement. A vacuum station includes two steel tanks, two vacuum pumps to induce the vacuum into the tank, and two pressure pumps to deliver the collected faeces to the biogas plant. Material and weight of other system parts is estimated.

Table 92: Material data for a vacuum station

Component	Material	Value	Unit	Quantity	Source
Vacuum tank (4 m³)	Sheet steel	1000	kg	2	Dehoust, 2005
	Excavation	16	m³		Estimation
Vacuum pump	Cast iron	100	kg	2	Bengtsson et al., 1997
Pressure pump	Cast iron	100	kg	2	Bengtsson et al., 1997
Small parts	Polyethylene	100	kg		Estimation

Biogas plant

The biogas plant for the co-digestion of faeces and biowaste consists of the following parts:

- Pretreatment of biowaste (“pulper”)
- Hygienisation tank
- Digester and gas storage
- Post-digestion tank
- Composting of digester residual in open piles
- Central heat and power plant for biogas usage
- service building

For the pretreatment and hygienisation, two tanks with 25 m³ each are provided, which can simultaneously act as buffer tanks. The residence time in the digester is set to 30 days on average, so that a daily load of ca. 20 m³ of digester substrate leads to a digester volume of 600 m³ (4x 150 m³). The post-digestion tank has a volume of 250 m³, and two gas storage tanks with 225 m³ each can store the generated biogas. The composting of the digester residual is done in roofed open piles, and the roof material is roughly estimated here. The two CHP engines (redundancy) have a maximum power output of 25 kW of electrical and 50 kW of thermal energy. For the housing of CHP engines, pretreatment processes etc, a service building (70 m²) is provided.

Table 93: Material data for a biogas plant

Component	Material	Value	Unit	Quantity	Source
Tanks (25 m³)	Concrete	10000	kg	2	According to Edelmann et al., 2001
	Steel, non-alloy	345	kg		
Pumps	Cast iron	100	kg	8	Bengtsson et al., 1997
Stirrer	Cast iron	150	kg	4	Estimation
Digester (4x 150 m³) incl. gas storage (2x 225 m³)	Concrete	202200	kg	1	Ronchetti et al., 2002
	Steel, non-alloy	4533	kg		
Post-digestion tank (250 m³)	Concrete	100000	kg	1	Edelmann et al., 2001
	Steel, non-alloy	3450	kg		
CHP engine (25 kW_{elec}, 50 kW_{therm})	Steel, alloy	118	kg	2	Ronchetti et al., 2002
	Polyethylene	2,6	kg		
Small parts	Polyethylene	1000	kg		Estimation
Service building (70 m²)	Concrete	112200	kg	1	Reckerzügl, 1997
	Steel, non-alloy	4100	kg		
Roof for open composting	Steel, alloy	5000	kg		Estimation

9.5 LCA model of a conventional wastewater treatment plant

9.5.1 Basic considerations

Concept

This process model is mainly based on linear input-output relations: substance and energy inputs are related to output via specific factors. The model describes the activated sludge process with nitrification and denitrification, considering typical parameters of German wastewater treatment plants (WWTP). Sludge treatment comprises of anaerobic digestion, dewatering and final co-incineration in a domestic waste incineration plant. Biogas from digestion is combusted in a central heating and power plant (CHP plant) to produce electrical and thermal energy.

The input substance flows are allocated to the output flows of discharged water, sludge and air specifically for each elemental flow. Material and energy demands of WWTP operation are calculated as well.

Product-specific relation

Municipal wastewater treatment plants treat wastewater from different origin (households, light and heavy industry, stormwater etc) which is mixed in the sewer. Due to the product relation of LCA, the model has to be capable to calculate the inputs and outputs of a specific wastewater being treated in the WWTP (problem of allocation in multi-input-processes). The present model tries to solve this allocation problem by considering causal relations between single wastewater components and emissions or input flows

Temporal scope

LCA models do not consider the time-related dynamic behaviour of the analysed systems. Hence, the present model is of a static type, calculating average loads and emissions.

9.5.2 Model description

9.5.2.1 Wastewater composition

The input wastewater is characterized concerning its elemental composition (COD, N, P etc). For the process model, the speciation of the elemental flows is important (e.g. the distribution of nitrogen on NH4-N, organic nitrogen, particulate nitrogen). Therefore, exemplary parameters describe the different fractions of the elemental flows according to Table 94. While particulate species can be separated physically by sedimentation, dissolved species have to be treated biologically (activated sludge) or chemically (precipitation).

Table 94: Assumed composition of influent wastewater

Species	Proportion [%]
Nitrogen	
NH4-N	54
NO3-N	0
N organic	35
N particulate	11
Phosphorus	
PO4-P	83
P particulate	17
Sulphur	
SO4-S	96
S particulate	4

9.5.2.2 Wastewater treatment

The present WWTP model includes the processes of primary, secondary and tertiary treatment as well as sludge thickening, anaerobic stabilisation and dewatering. The removal efficiency for COD, nitrogen, and phosphorus is adjustable via parameters (Table 95). The removal efficiency for nitrogen elimination is greater than 25%, due to the fact that around 25% of the nitrogen is usually incorporated into the excess sludge during biomass production. The model calculates the specific elemental allocation of wastewater components to the different output flows (discharged water, air, stabilized sludge, biogas) and the connected demand for chemicals of energy. In the following, the most important elements are discussed in detail.

Table 95: Parameters of LCA model for conventional WWTP

Parameters	Description	Value	Unit
ABBAU	Rate of decomposition of organic substance in stabilisation	50	%
CSBE	Efficiency of COD elimination	95	%
CVK	Proportion of COD eliminated in sedimentation	35	%
EG	Proportion of thermal energy supply from natural gas (rest is light oil)	50	%
ETH	Thermal energy demand per m³ of raw sludge	150	MJ/m³
FH	Static head of wastewater lifting at WWTP inlet	4	m
FHMEW	Demand of coagulation aid for dewatering	7	g/kg TS
FHMVD	Demand of coagulation aid in thickening	10	g/kg TS
NE	Efficiency of dissolved nitrogen removal (>25%)	80	%
PE	Efficiency of dissolved phosphorus removal	95	%
PFE	Proportion of phosphorus removal via precipitation (rest: biological P elimination)	50	%
T	Temperature	15	°C
TOC	Ratio TOC / COD	0,35	--
TRR	Dry matter content of thickened raw sludge, 2 < TRR < 5	5	%
TRST	Dry matter content of dewatered stabilised sludge Dewatering with centrifuge: 25 < TRST < 40	40	%
TTS	Sludge age	20	Tage
YC	Yield coefficient (g C biomass per g C substrate)	0,67	--

Carbon

Carbon content of influent and effluent is related to the chemical oxygen demand (COD). The relation of total organic carbon (TOC) and COD is usually between 0.28 and 0.4, in household wastewater it is typically around 0.35 (Zimmermann et al., 1996). The average elimination rate in German WWTP is ca. 93% for COD and 97% for biological oxygen demand (BOD) (ATV-DVWK, 2004). The particulate fraction of COD is separated by sedimentation in primary treatment, leading to its removal not in the activated sludge process, but in anaerobic sludge stabilisation where biogas production is increased. The dissolved COD is partly transformed into biomass in the secondary treatment (assimilation) or respirated to CO_2 and water (dissimilation). The ratio between assimilation and dissimilation is described by the yield coefficient Y, which is typically around 0.67 (ATV, 2000). The formation and endogenous decomposition of biomass is described by the following equation:

$$COD_{biomass} = COD_{decomposed} * \frac{Y}{(1 + b * t_{TS} * F_T)}$$

with: $COD_{biomass}$ = COD of formed biomass [mg/L]
$COD_{decomposed}$ = difference of COD in influent and effluent [mg/L]
$F_T = 1.072^{(T-15)}$
$Y = 0.67$ [g $C_{biomass}$/g $C_{decomposed}$]
$b = 0.17$ [d^{-1}] (at 15°C)
t_{TS} = sludge age [d]
T = temperature

The inert solids remaining from endogenous decomposition are estimated to account for 20% of decomposed biomass:

$$COD_{biomass,\ inert} = 0.2 * COD_{biomass} * t_{TS} * b * F_T$$

Neglecting the inert particulate influent COD, which is separated in primary treatment, the COD transfer in excess sludge is:

$$COD_{excess\ sludge} = COD_{biomass} + COD_{biomass,\ inert}$$

Nitrogen

Nitrogen content of WWTP influent consists of fractions of NH4-N, organic-bound nitrogen, and particulate nitrogen. The removal of total nitrogen amounts to 70 – 85% in average WWTPs if specific denitrification is employed. In average, 74% of total nitrogen load are eliminated in German WWTPs (ATV-DVWK, 2004). Around 20-30% of nitrogen are incorporated in biomass during microbial growth, and 5-10% are denitrified in anaerobic sludge of clarifier (Zimmermann et al., 1996). The particulate nitrogen is separated by sedimentation in primary treatment. The nitrogen speciation in effluent is assumed to 60% NO3-N, 20% N org., and 20% NH4-N.

Dissolved nitrogen is eliminated via incorporation in biomass or via denitrification to elemental N_2. During this process, minor emissions of N_2O and NH_3 arise. In this study, 0.3 % of influent nitrogen is transformed into N_2O and 0.6 % into NH_3 (adopted from Bardtke et al., 1994).

There are different concepts for a nitrogen balance of wastewater treatment. According to Scheer, 1998, the nitrified NH4-N ca be estimated as follows:

$$(\mathrm{NH4\text{-}N})_{\mathrm{NIT}} = \mathrm{TKN}_{\mathrm{influent}} - \mathrm{N}_{\mathrm{excess\ sludge}} - \mathrm{NH4\text{-}N}_{\mathrm{effluent}} - \mathrm{N\ org}_{\mathrm{effluent}}$$

Denitrified nitrogen is calculated by the following equation:

$$(\mathrm{NO3\text{-}N})_{\mathrm{DEN}} = (\mathrm{NH4\text{-}N})_{\mathrm{NIT}} + \mathrm{NO3\text{-}N}_{\mathrm{influent}} - \mathrm{NO3\text{-}N}_{\mathrm{effluent}}$$

For the calculation of nitrified and denitrified nitrogen, the fraction of nitrogen incorporated into the excess sludge has to be estimated:

$$\mathrm{N}_{\mathrm{excess\ sludge}} = 0.25 * \mathrm{TKN}$$

Phosphorus

Phosphorus content of wastewater consists of dissolved and particulate phosphorus species. It is assumed that particulate phosphorus is completely separated by sedimentation. Dissolved phosphorus can be eliminated by biological processes (incorporation in biomass or enhanced biological phosphorus removal “EBPR”) and chemical precipitation with ferric or alum salts. In modern WWTP with intended P elimination, the efficiency of P removal is 93 - 97% (ATV-DVWK, 2004). Biological phosphorus removal requires a special process engineering and favourable wastewater composition. In this study, biological P elimination accounts for only 20% of total P elimination. 80% of phosphorus elimination is done by addition of ferric chloride and subsequent precipitation.

Sulphur

Sulphur is present in wastewater as sulphate or particulate sulphur. It is assumed that particulate sulphur is separated in primary treatment and remains in sludge, whereas dissolved sulphate remains in the effluent except the proportion that is dissolved in sludge water content.

Heavy metals

Inorganic trace substances like heavy metals are removed in varying proportions from the wastewater and transferred into the sludge. The remaining heavy metals are discharged with the effluent. Transfer coefficients of heavy metals are compiled from various sources and listed in Table 96.

Table 96: Transfer coefficients of heavy metals in municipal WWTP

Element	% in effluent	% in sludge	Source
Lead	38 12 **here: 20**	62 88 **here: 80**	Raach et al., 1999 Fuchs et al., 2002
Cadmium	50 27 **here: 30**	50 73 **here: 70**	Koppe and Stozek, 1999 Fuchs et al., 2002
Chrome	50 15 **here: 20**	50 85 **here: 80**	Zimmermann et al., 1996 Fuchs et al., 2002
Copper	7 – 54 12 **here: 15**	46 – 93 88 **here: 85**	Overath et al., 1997 Fuchs et al., 2002
Nickel	60 37 **here: 40**	40 63 **here: 60**	Zimmermann et al., 1996 Fuchs et al., 2002
Mercury	9 30 21 **here: 20**	91 70 79 **here: 80**	Raach et al., 1999 Koppe and Stozek, 1999 Fuchs et al., 2002
Zinc	30 – 40 21 **here: 25**	60 - 70 79 **here: 75**	Koppe and Stozek, 1999 Fuchs et al., 2002

Inorganic salts

Most inorganic salts are highly soluble in water. Hence, it is estimated that they remain in the discharged water except the fractions contained in sludge water content. Chloride from addition of flocculants (ferric chloride) is added to the effluent. 43% of calcium and 5% of potassium are estimated to be bound in the sewage sludge

AOX

AOX is a sum parameter for adsorbable organic halogenated compounds. For this study, a transfer coefficient of 50% of AOX into the effluent is assumed (Koppe and Stozek, 1999). The remaining fraction of AOX is not metabolized, but absorbed to the sludge.

Sand and screenings

The further treatment of inorganic inert material (mainly sand) and screenings from primary treatment are neglected in this study.

Flocculation chemicals

Ferric (III) chloride ($FeCl_3$) is applied in the precipitation stage as flocculant. It is delivered as a 40% solution in water. Decisive for the applied dosage is the beta factor, which describes the molar ratio between eliminated phosphorus and applied iron. Usually, the dosage is in excess of regular stoichiometry. In this study, the beta factor is assumed to 1.5 (Scheer, 1998).

9.5.2.3 Sludge treatment

Raw sludge

Raw sludge is composed of the inert particulate substances from sedimentation, the generated biomass, and the remaining inert solids from endogenous decomposition of biomass (ATV, 2000). The contents of carbon, nitrogen, phosphorus, sulphur, iron, metals and salts are calculated via transfer coefficients from input loads. Beside the carbon content, organic substance is composed of oxygen and hydrogen. Their contributory fraction is calculated via general proportional factors of organic matter (oxygen: 70% of carbon, hydrogen: 15% of carbon).

For the conversion of COD of excess sludge into organic matter, the factor 1.45 g COD/g organic dry matter is applied. An increasing sludge age leads to more respiration of carbon into CO_2 and less organic matter in the excess sludge.

Thickening

The raw sludge is thickened by gravity in a settling tank with dosage of coagulation aids (poyacrylamide). A dose of 10 g /kg dry matter is assumed, leading to a final dry matter content of 5 % (Schumann et al., 1997).

Digestion

This model considers an anaerobic mesophile sludge digestion as described in Gujer, 1999. The thickened raw sludge is digested in a mesophile reactor (33-37°C) with a retention time of 15-30 days and converted into CO_2 and biogas. The process needs a thermal energy input of 130-180 MJ/m^3 raw sludge. A fraction of 40-50% of the organic matter content is converted to biogas, generating ca. 0.9 m^3 biogas per kg decomposed organic matter with an average density of 1.15 kg/m^3. The biogas contains 63-68 % CH_4, 32-37 % CO_2, 0-2 % N_2 and 0-1 % H_2S. Based on the average composition of the substrate, the hypothetical content of CH_4, CO_2, NH_3, and H_2S of the biogas can be calculated by the disproportioning equation (Tidden, 2003):

$$C_aH_bO_cN_dS_e + (4a-b-2c+3d+2e)H_2O \rightarrow$$
$$\rightarrow 1/8\,(4a+b-2c-3d-2e)CH_4 + 1/8(4a-b+2c+3d+2e)CO_2 + d\,NH_3 + e\,H_2S$$

The organic matter of raw sludge is composed of 49% starch, 38% proteins and 13% fats (Tidden, 2003). If this composition is assumed for the above equation, the proportional allocation of converted carbon can be estimated as follows:

$$100\%\ C_{raw\text{-}sludge,decomposed} \rightarrow 61\%\ CH_4\text{-}C \uparrow + 32\%\ CO_2\text{-}C \uparrow + 7\%\ HCO_3\text{-}C$$

The decomposition of proteins leads to the production of NH_4, which is recirculated via the sludge water to the wastewater influent. This results in an additional ammonia load for the nitrification stage. Depending on the efficiency of sludge dewatering, 10-15% of the influent ammonia load is from sludge water (Gujer, 1999). This model considers the recirculated ammonia load in terms of energy demand for the aeration process. Around 10% of the sulphur content of raw sludge is converted into H_2S (Raach et al., 1999)

Dewatering

The stabilised sludge is dewatered by a high performance centrifuge with addition of organic coagulation aids (polyacrylamide). A stabilised sludge with good properties can be dewatered to a final dry matter content of 35-40%. The dosage of polyacryamide is set to 7 g/g dry matter.

Biogas usage in CHP plant

The generated biogas is combusted in a central heat and power plant (CHP plant) to gain electrical and thermal energy. A potentially required gas conditioning prior to the combustion process is neglected in this study. The amount of biogas lost by accidental leakage is set to 0.75% according to Ronchetti et al., 2002. Furthermore, the burning of excess gas should be allowed for in cases of system malfunction or storage overflow. Ronchetti et al., 2002 assume that 5% of the gas volume is flared in a modern biogas plant if complete usage of biogas for energetic purposes is intended, which is adopted in this study. Emissions from biogas flaring are rated the same as during combustion in the CHP plant.

Although pilot injection gas engines may have a higher efficiency and lower capital costs, the CHP plant in this study is equipped with a spark-ignition engine ("otto engine") in lean combustion mode with high excess air. Thus, legal emission limits can be met more easily. Table 45 summarizes CHP parameters and respective emission factors.

Table 97: Parameters and emission factors for CHP plant

		This study	Biogas-CHP	Natural gas-CHP
Source			Ronchetti et al., 2002	Calculated from database (IFU and IFEU, 2004)
Engine		60 kW Lean burn engine	60 kW Lean burn engine	50 kW (elec) Catalysator engine
Efficiency		32% electrical, 57% thermal	32% electrical, 57% thermal	29,3% electrical 58,6% thermal
Emission	**Unit**			
CH_4, combustion	mg/MJ	2,5	2,48	3,78
CO_2	mg/MJ	(*)	81.308	55.151
NO_x (as NO_2)	mg/MJ	38	37,85	62,98
N_2O	mg/MJ	1,6	--	1,57
CO	mg/MJ	51	50,93	51,17
SO_x (as SO_2)	mg/MJ	30	29,91	0,43
NMVOC	mg/MJ	2,5	2,48	4,72
Dust	mg/MJ	1,6	--	1,57

(*) depending on CO_2 and CH_4-input, less CO

9.5.2.4 Energy demand

The energy demand of wastewater treatment plants has been extensively studied in literature (Müller et al., 1994; LfU, 1998). Many factors influence the actual energy demand of a certain WWTP. Beside the input-related parameters (COD and N load, influent volume), process layouts and the overall dimension of the plant can play a major role. Table 27 summarizes data from 1,097 investigated plants in Germany (LfU, 1998).

Table 98: Energy demand of municipal WWTP (LfU, 1998)

Relation	Median	80%-percentile	weighted average
kWh/(design-pe*a)	27,0	41,0	24,3
kWh/(pe * a)	41,5	64,0	31,7
kWh/m³	0,32	0,56	0,32
kWh/ (kg COD)	1,06	1,7	0,88

From the index figures of Müller et al., 1994 and LfU, 1998, specific energy coefficients can be derived according to Table 99. The calculation of the coefficients is based on specific data of wastewater volume, load and removal efficiency mentioned in the literature.

Table 99: Allocation of energy demand in WWTP

	Electricity	Thermal energy	Unit / allocation
Wastewater lifting facility	3.5		Wh/(m³ m pressure head)
Primary treatment			
Mechanical treatment and sedimentation	0.08		kWh/kg dry matter (primary sludge + sand)
Primary sludge pumps	6		Wh/(m^3 m pressure head)
Secondary treatment			
Aeration (3 m injection depth, 11% efficiency)	550		Wh/kg O_2
Internal circulation	0.01		kWh/m^3 wastewater
Recirculation and mixing	0.05		kWh/m^3 wastewater
Phosphate precipitation	0.37		kWh/kg P eliminated
Clarifier			
Clarifier	0.01		kWh/m^3 wastewater
Sludge treatment			
Sludge pumping	0.01		kWh/kg dry matter sludge
Thickening of raw sludge	0.03		kWh/kg dry matter raw sludge
Raw sludge and digester heating		130	MJ/m³ thickened raw sludge
Digester with mixing	0.12		kWh/kg dry matter raw sludge
Dewatering stabilised sludge (high performance centrifuge)	0.06		kWh/kg dry matter stab. sludge
Auxiliary			
Room heating, lighting etc	0.03	38	kWh/m³ wastewater kJ/m³ wastewater

Oxygen demand for carbon degradation and nitrification

The required oxygen demand for carbon degradation and endogenous respiration, nitrification and denitrification is listed in Table 100 according to ATV, 2000. A straight stochiometric calculation of nitrification and denitrification results in an oxygen demand for nitrification of 4.57 g O_2/g NH_4-N, while 62,5% of this oxygen demand is reclaimed during denitrification.

Table 100: Specific oxygen demand of biological treatment according to ATV, 2000

Process	Unit	Value or calculation
C-degradation and endogenous respiration	kg O_2/kg $COD_{respirated}$	$COD_{influent} - COD_{effluent} - COD_{excess\ sludge}$
Nitrification	kg O_2/kg $N_{nitrified}$	4,3
Denitrification	kg O_2/kg $N_{denitrified}$	- 2,9

For the calculation of the effective oxygen demand, it has to be considered that oxygen introduced into the aeration tank is only partially exploited by microorganisms. Hence, the oxygen input has to be significantly higher than the values above. Table 101 lists the assumptions for calculating the energy demand for aeration (Imhoff, 1990).

Table 101: Calculation of specific energy demand for aeration (according to Imhoff, 1990)

	Unit	Value
Specific energy demand (fine bubble aeration)	Wh/ (kg O_2 m)	20
Injection depth	m	3
Oxygen utilization at 3m injection depth	%	11
Specific energy demand for aeration		
C-degradation and endogenous respiration	Wh/kg $COD_{respirated}$	ca. 655
Nitrification	Wh/kg $N_{nitrified}$	2340
Denitrification	Wh/kg $N_{denitrified}$	- 1580

The following tables list the transfer coefficients of greywater treatment for both natural and technical processes in the composting scenarios. The co-treatment of faeces filtrate increase the nutrient loads to the greywater treatment process considerably, so that the transfer coefficients are slightly different from those of the vacuum scenarios.

Table 102: Transfer coefficients in greywater treatment with soil filter (*)

Input	Emission as	Effluent	Air	Reed	Filter material	Sludge
		[%]	[%]	[%]	[%]	[%]
Organic carbon	TOC	8.5				
	CO_2-C		80.7			
	TOC in sludge					10.8
N_{total}-N	NH4-N	2.6				
	NO3-N	32.9				
	organic N	20.8				
	NH_3-N		0.4			
	N_2O-N		0.1			
	N_2-N		36.2			
	N in reed			1.6		
	N in sludge					5.4
P_{total}-P	PO4-P	17.4				
	P in reed			2.2		
	P in filter material				44.5	
	P in sludge					45.9
K	K	97.3				
	K in reed			2.7		

(*) including sedimentation, soil filter, and post-precipitation, scenario COMP_NAT

Table 103: Transfer coefficients of elemental flows in greywater treatment with sequencing batch reactor*

Input	Emission as	Effluent	Air	Sludge
		[%]	[%]	[%]
TOC	TOC	6.2		
	CO_2-C		65.5	
	C_{org} in sludge			28.5
P_{total}-P	P-species	10.5		
	P in sludge			89.5
N_{total}-N	NH_4-N	2.6		
	NO_3-N	15.5		
	N-species	10.3		
	N_2-N		42.3	
	NH_3-N		0.4	
	N_2O-N		0.1	
	N in sludge			27.9

(*) including sedimentation, SBR, and sludge dewatering; scenario Comp_Tech

9.6 Substitution potential of secondary fertilizers from urine and faeces

This chapter describes the procedure to estimate the amount of industrial mineral fertilizer which can be substituted by urine and composted faeces and biowaste. First, some general boundary conditions and limitations are stated together with basic considerations of plant availability of nutrients.

Use of fertilizers in Germany

Table 104 shows the amounts of fertilizers that are applied on German agricultural soils on average [rounded estimation according to UBA, 2005].

Table 104: Average amounts of applied fertilizers on agricultural soil in Germany

Nutrient	Mineral fertilizer [kg/(ha a)]	Manure [kg/(ha a)]
N	110	200
P2O5	20	150
K2O	30	210
CaO	125	

In case of fertilizing with urine, it is assumed in the following that mineral fertilizer can be substituted by urine according to the nutrient equivalents. The remaining demand is met with manure. This assumption is only reasonable, if the specific advantages of mineral fertilizer are not required, i.e. the supply of selective nutrients available in short term during certain phases of plant growth. The short-term availability is of particular importance with nitrogen. Secondary fertilizers may cover the short-term nutritional requirement only partially, because they provide nutrients at a slower rate.

Additionally, it has to be noted that in Germany nitrogen and phosphorus are applied in excess of required amounts (115 kg N/(ha*a) and 23 kg P2O5/(ha*a), resp. [UBA, 2003]. Due to the aforementioned reasons, it is difficult to determine exactly the amount of mineral fertilizer that can be substituted by secondary fertilizers from urine, faeces and biowaste.

Plant availability

The plant availability of nutrients is decisive for the amount of mineral fertilizer that can be substituted by secondary fertilizers from urine, faeces and biowaste. Depending on the type of secondary fertilizer, considerable differences are observed in the fertilizing potential. Important factors influencing the fertilizing potential are for example the ratio C/N and C/P, the pH-value of the soil etc. More research is required to assess the plant availability of nutrients from secondary fertilizers. There are ongoing projects to investigate this issue [IME, 2005; SCST, 2006].

Basic definitions of plant availability of nutrients are explained in Herter and Külling, 2001:

- Utilisation: fraction of a nutrient application that is absorbed by a plant. In soils with high natural nutrient content, utilisation rate is small, because the plant is absorbing most of its required nutrients from the soil reserve.
- Nutritional effect: amount of crop surplus in comparison to crop without fertilizer application
- Availability: fraction of a nutrient application that is absorbed in the year of application (short-term) or over three years (medium-term). The long-term plant availability includes the transformation of non-available into available nutrients, which may take several years.

The availability can be roughly estimated from the solubility of the nutrients. A better estimation is achieved by the comparison of the relative utilisation or nutritional effect between secondary and mineral fertilizers.

Nitrogen is absorbed by plants mainly in the form of nitrate or ammonia. In organic fertilizers, nitrogen is predominantly bound to the organic matrix (on average, 60% in sewage sludge and more than 95% in compost) [Herter and Külling, 2001]. Therefore, the nitrogen content is not directly available to plants, but it is made partially available through microbial degradation over time.

Herter and Külling, 2001 are calculating the average availability of nitrogen from compost to 0-10% in the first year and 0-5% for the next year. The minor availability of nitrogen is unfavourable for the marketing of the compost, because nitrogen is often the limiting nutrient for crop yields.

If the soil is naturally providing sufficient amounts of phosphorus, the delayed release of phosphorus from a fertilizer does not negatively affect the crop yield [Herter and Külling, 2001]. For the assessment of secondary fertilizers, the long-term availability of phosphorus is seen as relevant. Herter and Külling, 2001 are assuming a long-term availability of 100% of the phosphorus content of compost for their calculation, similarly to Vogt et al., 2002.

Procedure for assessing the substitution potential

Nutrients from applied fertilizer can have the following fate [Finck, 1992]:

- they are plant-available and are absorbed by plants
- they are plant-available and are not absorbed by plants
- they are fixed in the soil matrix and are not plant-available in the long term
- they are lost from the root zone by emission to atmosphere or wash-out into deeper soil

The possible substance flows of fertilizers after application are depicted in Figure 62.

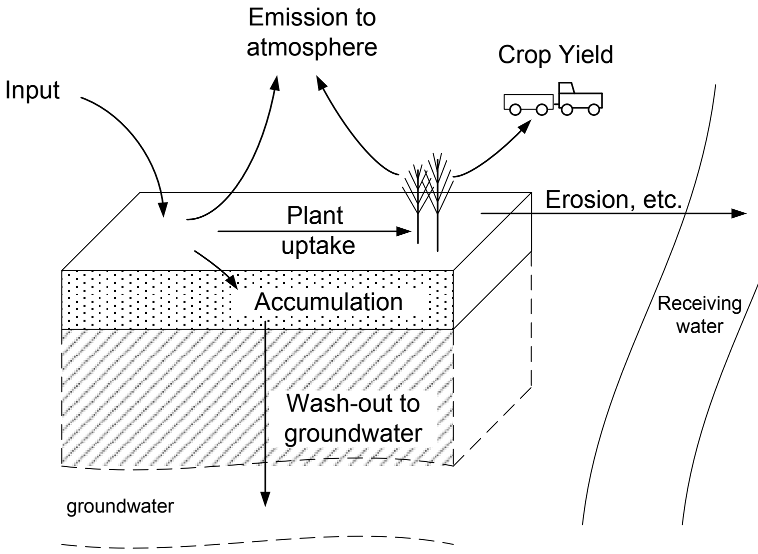

Figure 62: Pathways of nutrient flows after fertilizer application

The following equation 1 describes the plant-available nutrient fraction of a fertilizer:

$$N_{available} = N_{total} - N_{erosion} - N_{wash\text{-}out} - N_{atmosphere} - N_{fixed} \qquad (1)$$

where: $N_{available}$: plant-available fraction of nutrient
N_{total}: sum of all nutrient fractions before application
$N_{erosion}$: nutrient fraction lost by erosion
$N_{wash\text{-}out}$: nutrient fraction lost by wash-out into groundwater
$N_{atmosphere}$: nutrient fraction emitted into the atmosphere
N_{fixed}: nutrient fraction fixed to soil matrix (long-term

The amount of nutrients from mineral fertilizer wich actually can be substituted by secondary fertilizers is calculated via equation 2. The superior indices MF and SF indicate mineral fertilizer or secondary fertilizer.

$$\begin{aligned} N^{MF}_{total} = N^{SF}_{total} - N^{SF}_{erosion} + N^{MF}_{erosion} - \\ - N^{SF}_{wash-out} + N^{MF}_{wash-out} - N^{SF}_{atmosphere} + N^{MF}_{atmosphere} - N^{SF}_{fixed} + N^{MF}_{fixed} \end{aligned} \qquad (2)$$

The wash-out and erosion of nutrients are estimated to be equal for secondary and mineral fertilizers (in analogy to Tidaker, 2003). Thus, equation 2 is transformed into equation 3.

$$N^{MF}_{total} = N^{SF}_{total} - N^{SF}_{atmosphere} + N^{MF}_{atmosphere} - N^{SF}_{fixed} + N^{MF}_{fixed} \qquad (3)$$

For phosphorus and potassium, no losses from atmospheric emissions are relevant. XXX lists emission factors into the atmosphere and fixed fractions in soil. These factors are strongly dependent on pH-value, soil humidity and soil composition [ECETOC, 1994, EMEP/CORINAIR, 2004]. The application technology also has a great influence (drag hose, liquid injection, splash plate etc.). Factors in Table 105 are therefore simplified benchmark values.

Table 105: Data to estimate the plant availability of nitrogen for various types of fertilizers

Fertilizer	Emission	Fixation in soil matrix	Source
Mineral fertilizer (mean)	0,046 g NH3-N / g N 0,0125 g N2O-N / g N 0,007 g NO-N / g N	--	1) 1) 1)
Urine	0,06 g NH3-N / g N 0,0125 g N2O-N / g N 0,007 g NO-N / g N	--	3) 1) 1)
Compost	37% of NH4-N + 4% of none-NH4-N	0,9 g N / g N*	2) / 4) 2)
Digestor residue, stabilised	37% of NH4-N + 4% of none-NH4-N 0,0125 g N2O-N / g N	0,9 g N / g N*	2) / 4) 2)
Digester residue, not stabilised	0,22 NH3-N / g N 0,0125 g N2O-N / g N	0,9 g N / g N*	2) / 4) 2)

* after N emissions into the atmosphere; the calculation of the substitution potential of nitrogen is based upon the short- and medium-term plant availability

1) EMEP/CORINAIR, 2004
2) Assumptions in Vogt et al., 2002
3) Vinneras et al., 1998
4) Estimation considering the short- and medium-term availability, according to Herter and Külling, 2001

9.7 Composition and production of mineral fertilizers

9.7.1 Nutrient and heavy metal contents of mineral fertilizers

Commercial mineral fertilizers are single- or multi-nutrient fertilizers with variable nutrient content. The main nutrients nitrogen, phosphorus, potassium, lime and magnesium are usually specified in the form of N, P_2O_5, K_2O, CaO, and MgO. Table 106 and Table 107 show the most relevant types of mineral fertilizer together with their mean nutrient content and their market shares related to the nutrient content (Patyk and Reinhardt, 1997; Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001). The basis for the calculated market shares are the applied amounts of fertilizers in 1998/99 listed in Drescher-Hartung et al., 2001. Fertilizers with marginal market shares are neglected.

The multi-nutrient fertilizers NP-, NPK-, and PK-fertilizers are composed of defined parts of single nutrient fertilizers or contain chemical compounds with two types of nutrient (e.g. $(NH_4)_2HPO_4$). Lime fertilizer is assumed to be composed of 85% $CaCO_3$ and 15% CaO (Patyk and Reinhardt, 1997). As the most important K-fertilizer, potassium chloride (KCl) is considered. Magnesium is added to the production process of multi-nutrient fertilizers in the form of dolomite, and is then transformed to magnesium nitrate. However, the production of Mg-fertilizers is neglected in this study due to its minor contribution to the overall resource usage and heavy metal content of all fertilizers.

Table 106: Nutrient content and market shares of relevant mineral N- and P-fertilizers

Fertilizer	**Nutrient content [%]**		**Market share [%]**	
	N	P_2O_5	related to N	related to P_2O_5
Calciumammoniumnitrate	26,8		55	
Urea	46,7		12	
Ureaammoniumnitrate	32,0		18	
Singlesuperphosphate (18% P_2O_5)		20,0		2
Tripelsuperphosphate (50% P_2O_5)		48,5		11
Raw phosphate		26,0		3
NP-fertilizer, 20/20/0	20,0	20,0	7	29
NPK- fertilizer, 15/15/15	15,0	15,0	9	40
PK- fertilizer, 0/15/20		15,0		15

Table 107: Nutrient content and shares of lime and K-fertilizer

Fertilizer	**Nutrient content [%]**		**Share [%]**	
	K_2O	CaO	rel. to K	rel. to CaO
Potassium chloride	60,0		100	
$CaCO_3$ (lime)		54,3		85
Quick lime		97,0		15
Calciumammoniumnitrate (CAN)		21,0		--

When assessing the environmental impacts of mineral fertilizers, their heavy metal content is of particular importance. The partially high contamination of phosphate fertilizers with Cd, Cr, and Zn is well-known. Recent research results from the federal agency for agricultural research indicate considerable concentrations of uranium in certain fertilizers (Kratz, 2004; Fink, 2005). Uranium is enriched during the production process of superphosphate or triplesuperphosphate from raw phosphates from 13-75 mg/kg to 85-191 mg/kg due to its affinity to phosphorus. NP- or PK-fertilizers may contain up to 89-96 mg U/kg, and NPK-fertilizers still contain 14 mg U/kg. Mineral fertilizers without phosphorus content contain less than 1 mg U/kg. Assuming an average application of 22 kg mineral P-fertilizer per hectare of agricultural land, these concentrations result in a uranium load of 10-22 g/ha*a. Estimating the yearly loss due to erosion and crop yield to 1 g U/ha, an accumulation of toxic uranium in agricultural soils is likely.

Table 108: Concentrations of heavy metals and As in relevant mineral fertilizers

Fertilizer	**Concentrations of heavy metals and As [mg/kg dry matter]**								
	As [1)]	Cd [2)]	Cr [2)]	Cu [2)]	Ni [2)]	Hg [1)]	Pb [2)]	U [3)]	Zn [2)]
Calciumammoniumnitrate	3,30	0,25	8,70	4,00	3,80	0,01	21,40	< 1	38,30
Urea	0,04	0,13	0,50	0,50	0,70	0,01	0,60	< 1	1,90
Ureaammoniumnitrate		0,03	1,30	6,30	0,30		0,20	< 1	2,30
Singlesuperphosphate	3,70	10,80	114,00	17,20	28,80	0,02	18,50	138	236,00
Triplesuperphosphate	3,70	26,80	288,00	27,30	36,30	0,04	12,00	138	489,00
Raw phosphate	3,60	7,80	168,00	15,60	15,60	0,07	1,30	44	199,00
NP-fertilizer, 20/20/0		9,15	91,40	21,50	18,00	0,02	5,50	93	151,00
NPK-fertilizer, 15/15/15		3,78	45,80	11,30	10,90	0,06	14,80	14	116,00
PK-fertilizer, 0/15/20		7,98	191,00	19,30	19,90	0,08	14,40	93	152,00
Potassium chloride	0,01	0,08	3,50	2,90	1,50	0,02	0,50	< 1	3,70
Lime ($CaCO_3$)	0,05	0,30	7,50	8,20	6,10	0,04	5,90	< 1	41,20
Quick lime		0,10	19,20	11,10	6,00		2,80	< 1	15,80

[1)] Vogt et al. (2002) except NP-, NPK- and PK-fertilizer: Drescher-Hartung et al. (2001)

[2)] Drescher-Hartung et al. (2001)

[3)] Estimations according to Kratz (2004)

Table 108 lists the average heavy metal content of mineral fertilizers. Data is adopted from Drescher-Hartung et al., 2001 (Appendix 1A) and complemented by Vogt et al., 2002. The literature data is essentially based on the work of Boysen (Boysen, 1992). The values for uranium are mean values derived from the concentration ranges presented in Fink, 2005, and are therefore representing rough estimations. In general, the heavy metal content of fertilizers shows a wide variation limit due to the variable origin of raw materials.

For the calculation of mean heavy metal contents for the average N-, P-, K, and Ca-fertilizers (Table 109), the data from Table 108 is connected with the nutrient content and market shares of the respective fertilizers. Data for As is estimated for multi-nutrient fertilizers via single-nutrient fertilizers.

Table 109: Mean concentrations of heavy metals and As for average mineral fertilizers, related to the single nutrients

Values in mg/kg nutrient	As	Cd	Cr	Cu	Ni	Hg	Pb	U	Zn
N-fertilizer (as N)	9,3	6,0	77,9	26,0	20,9	0,07	54,9	51,5	203,0
P-fertilizer (as P_2O_5)	14,5	39,5	543,2	90,5	88,3	0,3	67,0	349,2	839,2
K-fertilizer (as K_2O)	0,1	0,1	5,8	4,8	2,5	0,03	0,8	1,0	6,2
Lime (as CaO)	0,1	0,5	14,7	14,6	10,5	0,06	9,7	1,0	66,9

9.7.2 Life cycle inventory of the production and supply of commercial mineral fertilizers

The production of mineral fertilizers and the associated substance and energy flows are documented in detail by Patyk and Reinhardt (Patyk and Reinhardt, 1997). However, emissions in surface waters are not considered within their study, although they can play an important role for the environmental evaluation (e.g. phosphate and fluoride emissions in processing of raw phosphates). Hence, aquatic emissions are adopted from a Suisse study (Gaillard et al., 1997) and recalculated, relating them to the average single-nutrient fertilizer via market shares and nutrient content. The Suisse study does not include all types of multi-nutrient fertilizers, so NP- and NPK-fertilizers are accounted for as ammonium-nitrate-phosphate and PK-fertilizers, single superphosphates and raw phosphates are calculated as triplesuperphosphate (conversion relative to the nutrient content).

Table 110 shows an abstract of important LCI data regarding the production process of mineral fertilizers. The data takes into account the complete processes of production and supply of mineral fertilizers, including transport and energy supply, starting with the extraction of resources until the packing of the marketable product.

Table 110: Life cycle inventories of mineral fertilizer production (abstract) from Patyk and Reinhardt, 1997 and Gaillard et al., 1997

Reference value		**1000 kg N**	**1000 kg P_2O_5**	**1000 kg K_2O**	**1000 kg CaO**
Input	unit				
Use of resources					
Primary energy carrier					
Lignite	kg	35	69	57	77
Natural gas	kg	881	180	173	17
Crude oil	kg	229	166	29	1,75
Hard coal	kg	107	62	40	15
Minerals and ores					
Raw potash	kg			10.500	
Limestone	kg	550			2.045
Raw phosphate ore	kg		4.060		
Sulphur	kg		272		
Cumulated energy demand (CED)					
CED (fossil)	MJ	48.264	16.337	9.866	1.842
CED (nuclear)	MJ	632	1.095	516	428
CED (regenerative)	MJ	178	278	79	14
Output					
Emissions (air)					
Particles	kg	0,10	0,51	0,05	0,60
Dust (>PM10)	kg	2,31	1,11	0,85	0,38
NH_3	kg	6,69	0,01	0,00	0,00
HCl	kg	0,07	0,02	0,07	0,01
N_2O	kg	15,05	0,04	0,05	0,03
HF	kg	0	0,023	0	0
CO2, fossil	kg	2.820 1)	1.117	617	343 2)
CO	kg	2,80	1,42	0,42	4,64
NOx	kg	15,76	8,58	1,15	0,24
SO_2	kg	5,16	11,98	0,27	0,07
VOC					
Benzo(a)pyrene	kg	7,07E-07	9,65E-07	2,01E-07	1,37E-08
Benzene	kg	0,01	0,01	0,00	0,00
Formaldehyde	kg	0,02	0,03	0,01	0,00
Methane	kg	7,45	2,07	1,38	0,31
NMVOC, unspecified	kg	0,54	0,49	0,12	0,01
PCDD, PCDF	kg	1,29E-09	2,38E-10	2,51E-10	3,74E-11

Reference value		1000 kg N	1000 kg P_2O_5	1000 kg K_2O	1000 kg CaO
Emissions (water)					
Metals					
Al	g	476,09	94,71	23,4	29,4
As	g	0,96	4,59	0,05	0,06
Cd	g	0,03	4,40	0,00	0,00
Co	g	0,95	0,19	0,05	0,06
Cr	g	4,94	23,04	0,28	0,30
Cu	g	2,40	22,47	0,12	0,15
Fe	g	334,19	146,89	33,10	119,00
Ni	g	2,43	18,11	0,12	0,15
Hg	g	0,00	4,18	0,00	0,00
Pb	g	2,67	19,58	0,19	0,18
Se	g	2,39	0,48	0,12	0,15
Zn	g	4,95	27,48	0,27	0,31
Nutrients					
Ammonia as NH_3	g	2,68	9,17	1,72	0,69
Nitrate as NO_3	g	189,15	8,16	1,20	0,75
Phosphate as PO_4	g	28,62	4400[3]	1,40	1,75
Salts					
Chloride	g	6219,00	5826,89	825,00	459,00
Cyanide	g	0,09	0,06	0,02	0,01
Fluoride	g	1,65	2200[3]	0,27	0,15
Sulphide	g	0,17	0,29	0,04	0,01
Hydrocarbons	g	0,027	0,05	0,007	0,005

[1] Including credit for the bonding of CO_2 in urea production; without credit (including CO_2 emissions by urea hydrolysis in soil: 2.980 kg if 10% market share of urea related to applied nitrogen

[2] For a fertilizing lime with the assumed main components (85% $CaCO_3$, 15% CaO), 670 kg CO_2 is being emitted in the soil per ton of applied CaO. These emissions are not included in this data, and therefore have to be considered during application of the lime

[3] Data for fluoride and phosphate emissions are estimated (see text)

Different production processes and countries of origin and considered in relation to their market share in Germany 1993 (aquatic emissions: 1998/99). Transport of the fertilizers to the field and emissions during application are not included, similarly the construction and maintenance of infrastructure is not included. Regarding the data quality, recent changes in market shares or production processes are not accounted for in this data set, except for the emissions of fluoride and phosphate during P-fertilizer production. In the multi-stage production process of P-fertilizer, considerable emissions of gaseous fluorides (HF, SiF_4) occur due to the high fluoride content of raw phosphates (2-5 %). In modern production facilities, these air-borne emissions are filtered from the exhaust air by gas scrubbers. However, uncontrolled fluoride emissions may occur from phosphogypsum stacks and wastewater ponds (e.g. in Florida (US), a pond water analysis revealed 9 g PO_4/L and 1 g F/L at pH 2.4 (Nifong, 1998)). Scarce literature data shows a wide range of emission factors for fluoride and phosphate emissions (Table 111) depending on country and environmental legislation. Hence, medium values are estimated for this study.

Table 111: Emissions of fluoride and phosphate in triple super phosphate production (mg/kg P)

	Fluoride		Phosphate	Remarks
Source	**Air**	**Water**	**Water**	
Audsley et al., 1997	460[1]	167000	138000	Data from 1992
Wiesenberger, 2002	2.7[2]	2680	4141	Data from Austria 2001
WMC 2003	3260[3]			Data from Australia 2003
EPA, 1995:	874[1]			Superphosphate production US
Emission factors	„site-specific“			Phosphogypsum stacks
HELCOM, 1996		690[1]	351	Recommendations for critical load
SCST	**10[2]**	**5000[1]**	**10000**	**estimated**

1) as fluorides 2) as HF 3) fluorides, calculated from production and emission data

9.8 Results

9.8.1 LCIA data for all impact categories

Indicator	Unit	WWTP	REF	COMP_NAT	COMP_TECH	VAC_NAT	VAC_TECH
CED fossil	MJ	2214745.98	4000865.56	3490261.14	3861089.04	2914514.76	3340889.34
CED nuclear	MJ	991677.04	1577264.21	1428146.47	1736040.04	1209113.62	1459425.30
ADP	kg Sb-eq	1258.82	2229.93	1955.37	2191.13	1630.25	1890.87
GWP	kg CO2 eq	302615.47	598213.15	547126.67	583336.76	424004.93	465850.81
AP	kg SO2-eq	1342.11	2319.17	3524.77	3515.73	2585.22	2530.83
EP-limP	kg PO4-eq	665.73	686.96	509.47	584.05	498.57	499.63
FAETP	kg p-DCB	20691.28	24033.05	23750.62	22882.95	21926.35	21238.62
TETP	kg p-DCB	10961.45	35363.33	15313.19	15025.51	12143.13	11873.09
HTP	kg p-DCB	43082.87	123196.36	59152.25	67732.41	42050.42	48783.97
AET (IMP2002)	kg TEG-eq	287064115.00	320810317.16	385513905.00	356592161.44	358413049.75	334800120.72
TET (IMP2002)	kg TEG-eq	164916878.24	223982159.79	306356594.54	297903640.07	256426095.75	250804345.19
EP (N + P)	kg PO4 eq	2616.72	2779.53	2520.66	1862.59	1611.09	1344.07

Table 112: Results of all LCIA indicators

9.8.2 Normalised LCIA data for all impact categories

Normalised indicator	Unit	WWTP	REF	COMP_NAT	COMP_TECH	VAC_NAT	VAC_TECH
CED fossil	pe * a	15.13	27.33	23.84	26.38	19.91	22.82
CED nuclear	pe * a	44.90	71.41	64.66	78.60	54.74	66.07
ADP*	pe * a	38.61	68.40	59.98	67.21	50.01	58.00
GWP	pe * a	24.64	48.72	44.56	47.51	34.53	37.94
AP	pe * a	91.93	158.85	241.42	240.80	177.07	173.34
EP-limP	pe * a	512.10	528.43	391.90	449.27	383.52	384.33
FAETP	pe * a	232.75	270.34	267.16	257.40	246.64	238.90
TETP	pe * a	156.37	504.47	218.45	214.34	173.23	169.37
HTP	pe * a	5.93	16.96	8.14	9.32	5.79	6.71
AET (IMP2002)	pe * a	166.84	186.45	224.06	207.25	208.31	194.58
TET (IMP2002)	pe * a	83.73	113.72	155.54	151.25	130.19	127.34
EP (N + P)	pe * a	402.57	427.62	387.79	286.55	247.86	206.78

* data for Western Europe 1995

Table 113: Normalised results of all LCIA indicators (Normalisation to Germany 2002)

Normalised indicator	Unit	WWTP	REF	COMP_NAT	COMP_TECH	VAC_NAT	VAC_TECH
CED fossil*	pe * a	15.13	27.33	23.84	26.38	19.91	22.82
CED nuclear*	pe * a	44.90	71.41	64.66	78.60	54.74	66.07
ADP	pe * a	38.61	68.40	59.98	67.21	50.01	58.00
GWP	pe * a	20.73	40.97	37.47	39.95	29.04	31.91
AP	pe * a	39.59	68.85	105.75	105.71	76.16	74.74
EP (N+P)	pe * a	77.63	89.12	93.22	76.03	61.40	54.00
FAETP	pe * a	13.35	15.51	15.32	14.76	14.15	13.70
TETP	pe * a	75.08	242.21	104.88	102.91	83.17	81.32
HTP	pe * a	1.85	5.29	2.54	2.91	1.80	2.09

* data for Germany 2002, not included in CML method

Table 114: Normalised results of all LCIA indicators (Normalisation to Western Europe 1995)

9.9 References

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9.10 List of Tables

9.11 List of Figures