Life Cycle Assessment of conventional and source-separation systems for urban wastewater management

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Dipl.-Ing. Christian Remy

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Promotionsausschuss:

Vorsitzender:	Prof. Dr.-Ing. Matthias Kraume
Gutachter:	Prof. Dr.-Ing. Martin Jekel
Gutachter:	Prof. Dr. rer.nat. Matthias Finkbeiner

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Vorwort

Nachhaltigkeit ist als Leitbild der zukünftigen Entwicklung in unserer Gesellschaft weithin anerkannt und wird von Wissenschaft, Öffentlichkeit und Politik zunehmend eingefordert. Damit ist auch im Bereich des Abwassermanagements eine Bewertung der bestehenden Systeme und eine Entwicklung hin zu nachhaltigeren Lösungen gefragt. Neben dem traditionellen System der Abwasserableitung und -behandlung wurden daher in den vergangenen Jahren neue Ansätze entwickelt, die eine kreislauforientierte Nutzung der im Abwasser vorhandenen Ressourcen ermöglichen. Die vorliegende Arbeit bietet nun einen systematischen Vergleich der ökologischen Nachhaltigkeit von konventionellen und neuen Systemen des kommunalen Abwassermanagements mittels der Ökobilanz-Methodik. Durch die Systemanalyse konnten die entscheidenden Vorteile der neuen Systeme nachgewiesen und damit Möglichkeiten zur weiteren Verbesserung der Nachhaltigkeit aufgezeigt werden.

Die vorliegende Arbeit entstand während meiner Tätigkeit als wissenschaftlicher Mitarbeiter am Fachgebiet Wasserreinhaltung der Technischen Universität Berlin. Sie basiert zu einem erheblichen Teil auf Ergebnissen des Projekts „Sanitary Concepts for Separate Treatment of Urine, Faeces and Greywater“ (SCST), finanziert vom Kompetenzzentrum Wasser Berlin gGmbH, Anjou Recherche (Paris) und der EU-Kommission (EU LIFE 03ENV/D/000025).

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Christian Remy

Zusammenfassung

Trennsysteme für die Behandlung von urbanem Abwasser erfassen die verschiedenen Abwasserteilströme separat und ermöglichen so die Rückgewinnung wertvoller Ressourcen (Energie, Nährstoffe) aus dem Abwasser. Daher werden sie allgemein als nachhaltiger im Vergleich zum konventionellen System der gemeinsamen Erfassung und Behandlung erachtet. Diese Hypothese wird in der vorliegenden Arbeit überprüft, indem die Umweltauswirkungen von konventionellem und stoffstrom-separierenden Systemen mit der Methodik der Ökobilanz (ISO 14040/44) verglichen werden. Für eine hypothetische Fallstudie eines Stadtgebiets mit 5000 Einwohnern werden zwölf verschiedene Szenarien für die integrierte Behandlung von Haushaltsabwasser und Bioabfall in einem Stoffstrommodel abgebildet. Die benötigten Sachbilanzdaten für alle relevanten Prozesse der Abwassererfassung und –behandlung sind aus Pilotprojekten und der Literatur zusammengestellt und werden durch qualifizierte Abschätzungen ergänzt. Sekundärfunktionen der Trennsysteme (Bereitstellung von Energie und Nährstoffen) werden berücksichtigt, indem das konventionelle System durch die entsprechenden Produktionprozesse für Netzstrom und Mineraldünger erweitert wird. Der Ressourcenverbrauch und die Emissionen werden für jedes Szenario aggregiert und in der Wirkungsabschätzung anhand von acht Indikatoren ausgewertet, darunter Energie- und Ressourcenverbrauch, globale Erwärmung, Eutrophierung, Versauerung sowie Human- und Ökotoxizität.

Die Ergebnisse der Wirkungsabschätzung zeigen, dass Trennsysteme signifikante Potentiale für ein nachhaltigeres Abwassermanagement bieten. Die Rückgewinnung von Energie aus Toilettenabwasser und vor allem Bioabfall in einem Vergärungsprozess kann den kumulierten Energieaufwand um bis zu 40% und die verbundenen Emissionen von Treibhausgasen um bis zu 46% reduzieren. Energetische Vorteile der Substitution von Mineraldünger sind relativ gering, aber die Qualität der organischen Dünger aus Urin und Fäkalien ist der von Mineraldünger oder Klärschlamm in Bezug auf Schwermetallgehalte überlegen. Das verbleibende Grauwasser kann in einem Belebtschlammverfahren mit geringerem Energieaufwand und besserer Ablaufqualität gereinigt werden als im konventionellen System. Eine naturnahe Reinigung des Grauwassers in Bodenfiltern senkt den Energieverbrauch erheblich, aber die eingeschränkte Entfernung von Phosphor kann hier das Eutrophierungspotential um bis zu 140% erhöhen. Grauwasser kann zudem für die Abwasserwiederverwendung in Membranbioreaktoren adequat gereinigt werden, obwohl die energetischen Vorteile der Wiederverwendung marginal sind. Bei der Ausbringung von flüssigen organischen Düngern aus Urin oder Fäkalien führen hohe Ammoniakemissionen zu einem um 60-110% erhöhten Versauerungspotential und sollten daher durch geeignete Ausbringungstechniken minimiert werden.

Insgesamt zeigt die Gruppierung und Wichtung der Indikatoren signifikante Vorteile von Trennsystemen in Bezug auf ökologische Nachhaltigkeit. Dennoch ist die Auswahl einer geeigneten Prozesskombination für Trennsysteme essentiell, um diese Vorteile zu realisieren, da das konventionelle System in Bezug auf Energieverbrauch und Ablaufqualität bereits optimiert wurde. Durch Sensitivitätsanalysen wurden entscheidende Schlüsselparameter der Sachbilanz identifiziert. Funktionelle Definitionen und die Auswahl sowohl der Indikatoren zur Wirkungsabschätzung als auch der Bewertungsmethode können die Ergebnisse der Ökobilanz erheblich beeinflussen.

Abstract

Source-separation systems for urban wastewater management collect the different wastewater flows separately to facilitate the recovery of valuable resources from wastewater (energy, nutrients). Thus, they are claimed to be more sustainable than the conventional concept of combined drainage and treatment. This hypothesis is verified in this study by comparing the environmental impacts of conventional and source-separation systems with the methodology of Life Cycle Assessment (ISO 14040/44). In a hypothetical case study for an urban area (5000 inhabitants), twelve different scenarios for the integrated management of household wastewater and biowaste are set up in a substance flow model. Required inventory data for all relevant core processes of wastewater collection and treatment is compiled from pilot projects and literature and is complemented by qualified assumptions. Secondary functions of separation systems (supply of energy and nutrients) are considered by expanding the conventional system with the respective production processes for grid energy and mineral fertilizer. Resource demand and emissions are aggregated for each scenario and evaluated in Life Cycle Impact Assessment with a set of eight indicators for energy and resource demand, global warming, eutrophication, acidification, and human and ecotoxicity.

Results of the impact assessment show that separation systems offer significant potentials for an increase in sustainability. Recovering energy from the organic matter of toilet wastewater and especially biowaste in a digestion process can decrease the cumulative energy demand by up to 40% and related emissions of greenhouse gases by up to 46%. Energetic benefits of mineral fertilizer substitution are relatively low, but the quality of organic fertilizers from urine and faeces is superior to mineral fertilizer or sewage sludge in terms of lower heavy metal content. The remaining greywater can be treated in an activated sludge process with less energy demand and better effluent quality than in the conventional system. Natural treatment in soil filters can further reduce the energy demand considerably, but the insufficient retention of phosphorus in soil filters can seriously increase the eutrophication potential by up to 140%. Greywater can also be adequately treated for non-potable reuse with membrane bioreactors, although the energetic benefits of wastewater reuse are marginal. During the application of liquid organic fertilizers from urine and faeces, increased emissions of ammonia lead to a higher potential for acidification (+ 60-110%) and should be minimized by adequate application techniques.

Overall, grouping and weighting of the indicators reveal significant benefits in ecological sustainability for separation systems. However, the choice of an appropriate combination of process technology for separation systems is essential for a realization of these potential benefits, because the conventional system has already been optimized in terms of energy demand and effluent quality. In sensitivity analysis, decisive key parameters of the inventory are identified. Functional definitions and the choice of both indicators for impact assessment and valuation methods can have a considerable impact on the results of this LCA.

Contents

1 Introduction

„Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs"
The Brundtland Report (WCED, 1987)

The concept of sustainable development has been postulated by the United Nations in 1987 in view of the rising pressure on natural resources and ecosystems caused by human activities. Thus, the global community recognized the limitation of global resources and the need for a paradigm shift in development towards sustainability. First announced by the Club of Rome in 1973 (Meadows et al., 1973), the limits of growth have since then become obvious in many fields of human activities, e.g. regarding the depletion of resources such as fossil fuels, agricultural land, or local availability of freshwater. Continuous growth of global population, rapid urbanisation, and the consequences of climate change will certainly put additional pressure on these resources in the future, further increasing the need for sustainable development (UN, 2008).

Hence, the need to design and operate sustainable systems in society has also been recognized for the management and disposal of municipal wastewater (Wilderer, 2004). Traditionally, the functions of urban wastewater management are the protection of human health by ensuring the safe disposal of wastewater and the protection of aquatic ecosystems from negative impacts of effluent discharge. The latter function also includes the protection of surface waters which serve as drinking water supply. These functions can be adequately fulfilled with the historically grown concept of wastewater collection and treatment as it is practised in industrialized countries. This concept is based on the combined collection of municipal wastewater in a flushing sewer and its treatment in a centralized wastewater treatment plant prior to the discharge into surface waters (hereafter denoted as "the conventional system"). However, the sustainability of this concept has been questioned within the scientific community (Wilderer, 2004). To further elaborate this issue, a closer look at the historical development of the conventional system of wastewater management can be helpful.

Historical development of urban wastewater management

Historically developed in central Europe in the second half of the 19th century, open drainage channels were built targeting the disposal of wastewater outside of the urban area without hygienic risks. The concept of flushing away all wastewater flows and pathogens with large amounts of water was originally designed to prevent the inner-city outbreaks of water-borne diseases such as cholera and typhus, causing a severe amount of deaths in regular intervals at that time. The immediate success in improving the

hygienic situation of the urban population led to the rapid implementation of flushing sewers in urban and also rural areas of industrialized countries in the 20th century.

Thus, the linear disposal-oriented system of combined collection of wastewater in a flushing sewer (Figure 1) has survived many decades and is still the guiding principle of urban sanitation today. Additionally, the need for a protection of receiving surface waters from negative impacts of wastewater discharge led to the development of sophisticated treatment processes for the combined wastewater, targeting the removal of solids, organic matter, and nutrients. In Germany, conventional wastewater treatment plants (WWTP) usually include a combination of physical, biological and chemical processes to ensure the compliance with stringent effluent standards. Recently, the question of implementing additional treatment stages has been raised for improving the hygienic quality of the effluent or removing potentially harmful micropollutants such as pharmaceuticals or endocrine disruptors (DWA, 2008a).

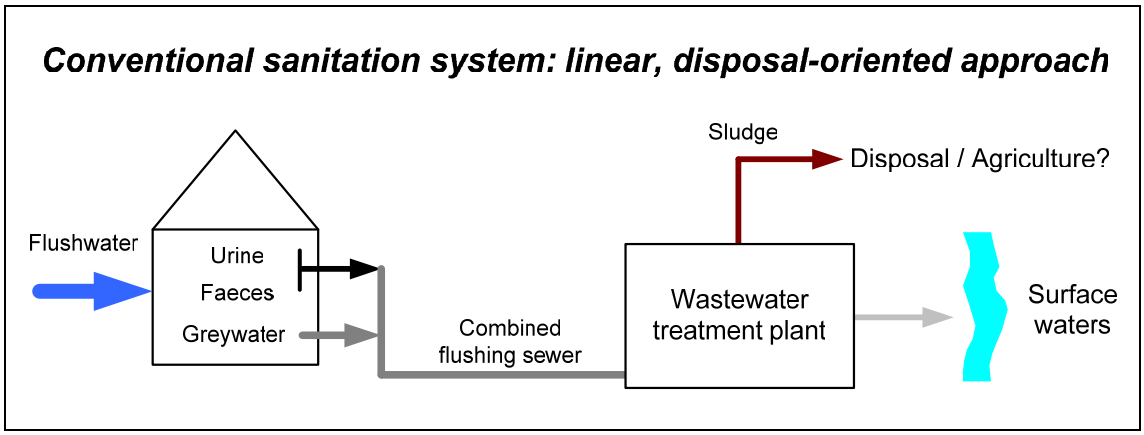

Figure 1: Conventional system of urban wastewater management

Inherent drawbacks of the traditional approach

Despite its successful history, the conventional wastewater system is originally designed as a linear disposal-oriented concept. Its focus is the safe disposal of wastewater while minimizing negative impacts on receiving surface waters, thus being a typical "end-of-pipe" technology. In terms of sustainability, the conventional system has been questioned for its capability to meet future requirements for sustainable system design. In particular, the following drawbacks have been identified for the conventional system (Wilderer, 2005):

- The use of freshwater to transport human excreta in large flushing sewers leads to a high drinking water demand. In water-scarce regions, additional pressure is put on limited freshwater resources.

- Organic matter and nutrients have to be eliminated in the WWTP with a considerable effort in energy and chemicals to protect the receiving surface waters.
- The recovery potential of valuable constituents of wastewater such as organic matter and nutrients is limited to the application of sewage sludge in agriculture. However, this option is discussed controversially due to the function of sewage sludge as a pollutant sink (Hartmann et al., 2004). Other options for nutrient recovery are being proposed, but their large-scale implementation is still pending and can be energy-intensive (Cornel and Schaum, 2009).
- The reuse of treated wastewater for non-potable purposes requires an extensive treatment to comply with the relevant standards, especially concerning hygienic quality and nutrient content (DWA, 2008b).
- Conventional wastewater collection and treatment requires large investments in relatively inflexible infrastructure. In industrialized countries, existing sewers have to be maintained and renewed with considerable financial effort, whereas the installation of completely new systems in developing or threshold countries may be delayed or entirely inhibited by the high investment needs.

The need for sustainable solutions for urban wastewater management

Hence, the sustainability of the conventional approach of urban sanitation is challenged with regard to high resource demand, limited recovery potential for valuable resources, and large investment needs. However, the conventional system has already been fully implemented in most industrialized countries: 95% of the population is connected to a public sewer system in Germany (DESTATIS, 2007). Nevertheless, considerations to improve the sustainability of wastewater management are necessary for a slow but constant shift towards a sustainable society. Existing systems have to be maintained or reconstructed: 20% of the existing urban sewer systems in Germany have been identified as requiring restoration in the next years, with an estimated investment of 53 billion Euro (Berger and Lohaus, 2005). Here lies an opportunity to gradually improve the sustainability of our wastewater management systems.

On a global scale, the situation is much more pressing. Many developing and threshold countries do not have sufficient coverage of urban areas with sanitation systems at all, with 2.4 billion people living without adequate sanitation today (WHO/UNICEF, 2008). Thus, large efforts are being undertaken to build up sanitation systems in these countries, with a focus on rapidly growing urban areas. Providing adequate sanitation for the global population is one of the Millennium Development Goals of the United Nations (UN, 2000). Without adequate sanitation, the negative impacts on human health (e.g. through water-borne diseases), economic development and ecosystems (e.g.

hypoxic zones in coastal areas (Diaz and Rosenberg, 2008)) are obvious in the developing world (UN-WWAP, 2003).

In all, there is a rising demand for sustainable systems for urban wastewater management, more distinct and pressing in developing countries, but also relevant for industrialized societies. Having identified the inherent drawbacks of the conventional disposal-oriented approach, several research groups started to develop future concepts for urban sanitation in the 1990s, targeting the recovery of resources from municipal wastewater (Larsen and Gujer, 1996; Hanaeus et al., 1997; Otterpohl et al., 1997).

Source separation systems

The difficulty of recovering potentially valuable resources from wastewater mainly originates from the commingling of wastewater flows with different characteristics. The exemplary distribution of nutrients in urine, faeces, and greywater (i.e. wastewater from kitchen, bathrooms, washing etc) shows that nutrients are mainly contained in human excreta, whereas greywater has low nutrient content in a high volume (Figure 2). By mixing these different wastewater flows, a high dilution of nutrients and organic matter occurs, making their recovery a difficult task. Hence, the basic principle of recovery-oriented systems is the separate collection of the different wastewater flows at the source ("source separation") (Figure 3).

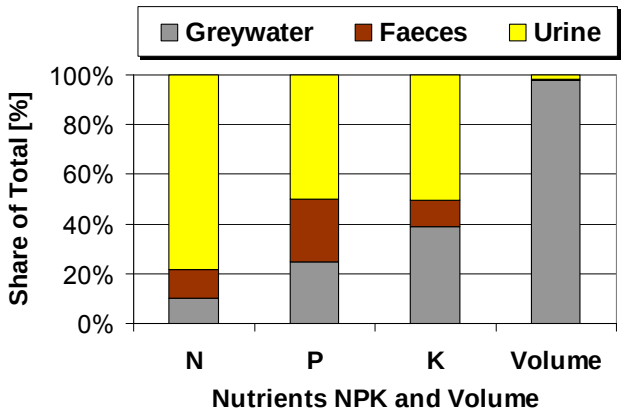

Figure 2: Volume and distribution of nutrients in urine, faeces, and greywater

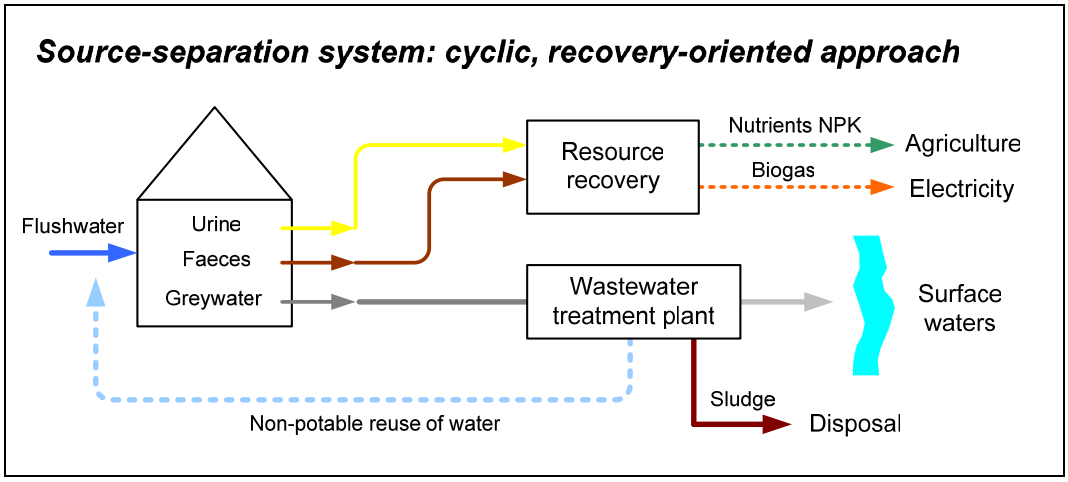

Figure 3: Source separation system for urban wastewater management

If the different wastewater flows are separately collected, resource recovery from highly concentrated flows (urine and faeces = toilet wastewater) is achievable with a reasonable effort. Separation systems can include various combinations of available technology for the different flows (Otterpohl et al., 1999; Wilderer, 2004). For example, the following features can be part of a separation system:

- The use of **vacuum drainage** for urine and faeces enables their transport with low amounts of flushwater, resulting in low dilution of the concentrated flows.
- With **urine separation** toilets, undiluted urine ("yellow water") can be collected separately from faeces and flushwater ("brown water"). Thus, urine can be directly applied in agriculture as a valuable fertilizer.
- Organic matter from faeces can be converted to biogas in a **digestion** process, recovering a substantial part of the energy bound in the organic matter. In an integrated approach for waste handling, biowaste from households can be easily added to the biogas plant. The residual sludge of the digestion process still contains valuable nutrients and can be applied as fertilizer in agriculture.
- Faeces and biowaste can also be treated in a **composting** process to obtain a valuable soil conditioner with some nutrients. Therefore, flushwater has to be separated from faeces beforehand to maintain aerobic conditions during composting.
- The remaining greywater – high in volume, but low in nutrient content – can be easily treated in a conventional WWTP. Moreover, it can be treated adequately in a low-energy natural process with a planted **soil filter** ("constructed wetlands"). Finally, a sufficient degree of purification for **non-potable reuse** of treated greywater can be reached with a **membrane bioreactor** (MBR).

The concept of source separation is not new, but has only recently been reconsidered for modern urban wastewater management. Following its rediscovery in the 1990s, multiple research projects targeted the development and optimization of source separation (e.g. Stockholm Vatten, 2000; Oldenburg et al., 2002; Otterpohl and Oldenburg, 2002; Peter-Frohlich et al., 2007; Larsen and Lienert, 2007). Several pilot plants were built to gain experience in installation, operation and maintenance of the different components of separation systems. Despite the positive results of the pilot projects, large-scale implementation of separation systems in industrialized countries is still pending. Nevertheless, the possibility for a paradigm shift in wastewater handling towards source separation is being intensively discussed by researchers and practitioners (DWA, 2008c; Larsen et al., 2009; Guest et al., 2009).

Assessing environmental sustainability with Life Cycle Assessment

As shown above, separation systems are usually claimed to be more sustainable than the conventional sanitation system, a thesis often supported with conceptual benefits and qualitative arguments. Quantitative evidence of the sustainability of separation systems is scarce. Evaluating the economic sustainability of different options for urban sanitation is relatively easy, because costs for installation, operation and maintenance can be directly compared between the systems. Quantifying environmental impacts for a comparison between conventional and separation systems is a more difficult task.

A suitable tool for the quantification of environmental impacts of a technical system is a method called “Life Cycle Assessment” (LCA), a methodology originally developed in the 1970s for assessing the environmental impacts of industrial products (Siegenthaler, 2006). Meanwhile, LCA has become a widely applied method for sustainability assessment of products or services (Klöpffer and Grahl, 2009). Rules for its application are precisely regulated in ISO standards 14040/44 (ISO 14040, 2006; ISO 14044, 2006), which have been updated recently (Finkbeiner et al., 2006). The LCA approach is characterized by taking a “life cycle perspective”: the assessment includes all relevant processes for the product or service under investigation “from the cradle to the grave”, following the entire life cycle. Additionally, all results are related to a functional unit, allowing the quantitative comparison of environmental impacts of different systems which provide the same function. Thus, LCA is an appropriate tool for a quantitative comparison of two or more technical systems in their environmental impacts.

Goals of this thesis

The present thesis aims at a quantitative comparison of environmental impacts of conventional and separation systems for urban sanitation in an industrialized environment. Thus, the ecological sustainability of separation systems shall be verified in a reproducible and transparent manner. Therefore, the study follows the methodology of Life Cycle Assessment as defined in ISO 14040/44. During the assessment, potential hotspots of separation systems in terms of environmental impacts shall be identified for further optimization of processes and system layout. Finally, key parameters for the environmental comparison shall be identified for a simplification of future LCA studies in this field. In particular, the following questions shall be discussed:

- Is source-separation more sustainable in terms of environmental impacts than a conventional system of combined drainage and treatment?
- What are the decisive benefits of separation systems?
- Which environmental drawbacks of separation systems can be identified, and how could they be minimized?
- Which are the important key parameters for the impact assessment?

Structure of this study

This study is structured according to the requirements of LCA as defined in ISO 14040/44 except for the critical review (ISO 14040, 2006; ISO 14044, 2006). After a review of relevant literature in the field of sustainability assessment of wastewater systems, the approach of this study is precisely illustrated (chapter 2). In the subsequent chapters, the various parts of LCA are performed for a hypothetical case study. In detail, this includes the following steps (Figure 4):

- **Definition of goal and scope** (chapter 3), including system functions, scenarios, system boundaries, and methodological issues of LCI and LCIA
- **Life Cycle Inventory or LCI** (chapter 4), documenting the inventory data for all relevant processes which is fed into a substance flow model
- **Life Cycle Impact Assessment or LCIA** (chapter 5), calculating the environmental impacts of aggregated emissions and resource demand from LCI
- **Interpretation** of results (chapter 6), including a discussion of relevant outcomes with special regard to data quality and sensitivity analysis.

The thesis is completed by summarizing the results of this case study and presenting relevant conclusions (chapter 7). Additional information (e.g. sources for inventory data, model descriptions, and numerical results of inventory analysis) is documented in the annex of this thesis.

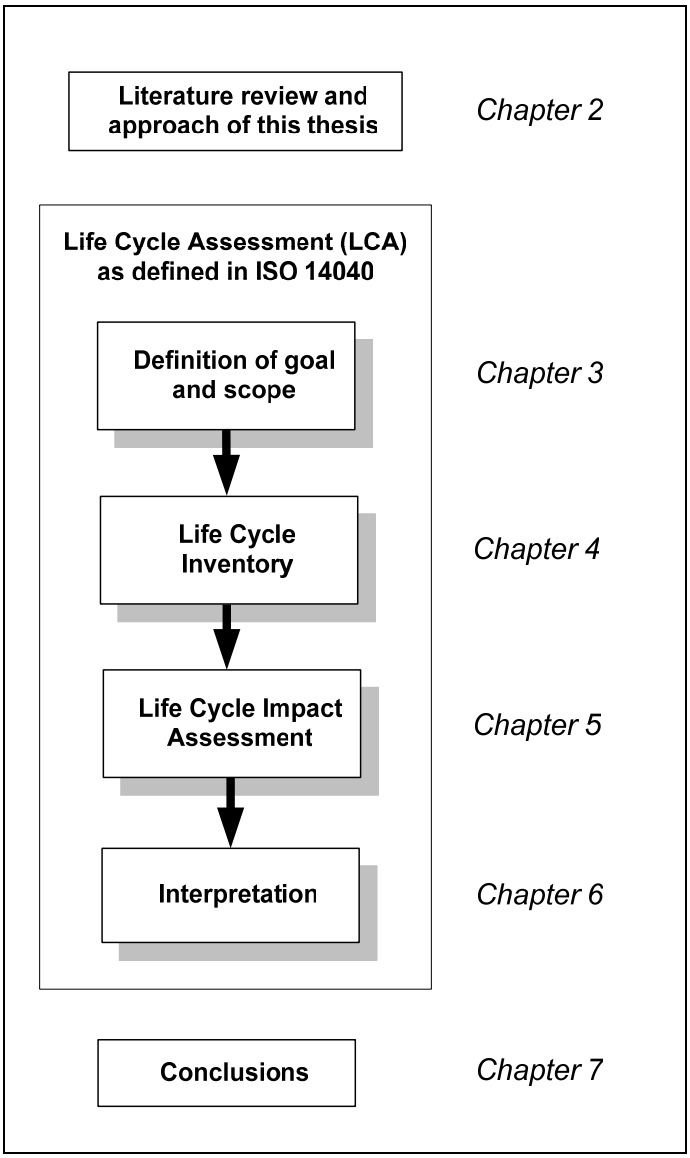

Figure 4: Structure of this thesis

2 Literature review and approach of this study

2.1 Sustainability assessment in water management

The rising pressure on limited global resources and ecosystems through anthropogenic activities has raised awareness of the need to design and operate sustainable systems. In the field of water management, numerous studies have been published which aim at assessing the sustainability of water supply and wastewater treatment options

Development of methods for sustainability assessment

Several attempts of developing a consistent methodology for sustainability assessment of water systems have been described in literature. Some of these studies include all three dimensions of sustainability, i.e. ecological, economic, and social issues, also known as "triple bottom line" (e.g. Ellis and Tang, 1990; Otterpohl et al., 1997; Hellström et al., 2000; Steinberg et al., 2002; Balkema, 2003). While evaluating ecological, economic and social issues, the high number of indicators (up to 35 different criteria) often complicates a conclusive comparison between two different systems. A carefully defined procedure for weighting the indicators against each other is required to come to a distinct result. An overview of relevant studies and sustainability indicators developed for water systems can be found in the thesis of Huegel and Balkema (Huegel, 2000; Balkema, 2003).

Other studies focus on the environmental performance of water systems, developing specific sets of environmental indicators applicable in the water sector (e.g. Lundin and Morrison, 2002; Mühleck et al., 2003). These environmental impacts are often characterized following the methodology of LCA, i.e. adopting the life cycle perspective and broadening the system boundaries to include associated background processes. In addition to LCA specific impact indicators, other types of indicators on the level of LCI are introduced here, e.g. energy demand, total water use, or recycling rate. Again, systematic difficulties arise while aggregating the different indicator results into a conclusive outcome.

Some authors finally deduct a numeric decision support tool from their sustainability assessment to facilitate the choice of an optimum solution under certain conditions (Ellis and Tang, 1990 ; Balkema et al., 2001; Norstrom et al., 2008). However, boundary conditions of water systems have a high influence on the choice for an appropriate solution and are very much depending on local conditions. Thus, the development of a universal decision support tool covering a wider area of application is a daunting task.

Life Cycle Assessment for wastewater management

The use of Life Cycle Assessment (ISO 14040, 2006) as an appropriate tool for the environmental assessment of wastewater management systems has been systematically evaluated by Huegel (Huegel, 2000). On the basis of four case studies, Huegel investigates methodological issues of LCA for four different targets of study: WWTP as downstream process in other LCA, comparative LCA for optimization of WWTP, ecological efficiency of sub-processes in WWTP, and strategic planning of wastewater management. The author identifies systematic problems in the methodology of LCA, particularly for the definition of functional unit, impact assessment and data management:

- Although the defined function (= wastewater treatment) should be comparable between different scenarios, different wastewater treatment technologies can differ in the treatment efficiency (= effluent loads) that they deliver. Thus, they may not provide exactly the same function and are per definition not comparable with LCA. The careful definition of system functions is necessary for a valid comparison between different scenarios.
- In impact assessment, short-term, acute and local impacts are usually omitted in LCA methodology, even though a major task of wastewater treatment is to control in particular short-term, acute and local emissions.
- The extensive amount of data required for assessing a complete wastewater system complicates the setup of the Life Cycle Inventory, often lacking high-quality data for certain processes or infrastructure.

Nevertheless, LCA is seen as a useful tool for a systematic investigation of the environmental impacts of wastewater systems (Huegel, 2000). For LCAs designed for strategic planning of wastewater management, further development of impact assessment and evaluation procedure is strongly recommended based on the available methods at that time.

Case studies

Aside from methodological improvement of sustainability assessment, a number of case studies have been carried out to compare different scenarios of water and wastewater management in their impacts on the environment. These studies are often limited in their system boundaries to a specific sub-part of the overall water management system, e.g. drinking water supply or installations (Crettaz et al., 1999, Gabriel and Kreissig, 2006; Stokes and Horvath, 2006; Barrios et al., 2008), wastewater treatment (Roeleveld et al., 1997; Hospido et al., 2004; Hoibye et al., 2008; Wenzel et al., 2008), or sludge disposal (Dennison et al., 1998; IFEU, 2002; Schubert, 2006). Studies which include the

complete water management system or even the water cycle (e.g. with catchment areas) are scarce due to the complexity of the system and the high amount of data required (Mühleck et al., 2003; Lundie et al., 2004; Lassaux et al., 2007). Separation systems have also been assessed with LCA, and the following chapter describes a selection of relevant case studies.

2.2 Review of LCA case studies for wastewater treatment

The methodology of LCA has already been applied in previous research since the mid 1990s to study the environmental impacts of different options for wastewater treatment. Several studies have been carried out to compare different scenarios of wastewater management, including both conventional and separation systems. However, the existing studies differ widely from each other in terms of layout and scale of the investigated systems, system boundaries, level of detail for Life Cycle Inventory, and the applied methodology of impact assessment (Table 1). In the following, the most important LCA case studies in the field of wastewater treatment are shortly summarized in chronological order.

Bengtsson et al 1997

This study evaluates different options for wastewater and sludge treatment for three case studies in Sweden with the LCA methodology (Bengtsson et al., 1997). Wastewater treatment options include urine separation or liquid aerobic composting of blackwater and organic kitchen waste, both with nutrient recycling to agriculture. The substitution of mineral fertilizer is accounted for in the LCA. Both separation systems show less environmental impacts than the conventional system, mainly due to a reduction of water emissions from the wastewater treatment plant and a conservation of fossil fuels due to the substitution of mineral fertilizer. In another publication, the authors emphasize the influence of system boundaries and scale on the calculated environmental loads (Lundin et al., 2000). Large economies of scale in environmental terms can be gained both for the operation and construction phase (smaller systems have higher relative impacts in relation to larger systems). Some of the most important environmental benefits of separation systems emerge only when system boundaries include associated processes such as substituted mineral fertilizer production.

Roeleveld et al 1997

Roeleveld and co-workers use the LCA method to quantify the environmental impacts of different options for conventional wastewater treatment in the Netherlands (Roeleveld et al., 1997). The scenarios cover an increasing degree of treatment, starting

with the removal of organic matter, and adding nitrification, denitrification, biological and chemical phosphorus elimination, and finally tertiary filtration plus activated carbon. Sludge handling and disposal is not considered within this study. With increasing nutrient removal, the contribution of the wastewater treatment process to the total environmental impacts in the Netherlands is relatively low. Eutrophication and aquatic ecotoxicity are the decisive indicators after normalisation. Consequently, the authors emphasize the importance of effluent-derived impacts from nutrients, heavy metals and organic pesticides, whereas the energy consumption of the process is of minor relevance. Likewise, the construction of the plant itself and the production of auxiliary chemicals for its operation are not decisive for the environmental performance in this LCA.

Tillman et al 1998

Tillman and co-workers apply the LCA instrument to municipal planning of wastewater systems in two Swedish settlements, a suburban area (12,000 inhabitants) and a coastal village (900 inhabitants) (Tillman et al., 1998). They compare the existing conventional wastewater systems with a separation system consisting of urine separation and application in agriculture, faeces co-digestion with biowaste, and greywater treatment in sand filter beds. The authors distinguish a *core system* of wastewater collection, treatment, and sludge disposal, and an *enlarged system* where effects on the surrounding technical systems are taken into account (e.g. energy production, drinking water production, fertilizer substitution). Whereas the core system can be modelled with higher accuracy, the enlarged system is more relevant for the assessment of the overall environmental impacts, although it is naturally based on data with higher uncertainty. For the core system, the study includes the construction expenditures. The separation scenario requires more electricity and fossil fuels for the construction than the conventional system, mostly due to the collection tanks for urine separation. Results for the enlarged system show that the separation scenario can save considerable amounts of energetic resources in the small village. The suburban area has a heat recovery system from the wastewater, which makes it energetically favourable to the separation scenario (existing system is a net producer of energy). The nitrogen emissions to surface waters are substantially reduced by urine separation in both settlements. Through different weighting procedures, the authors identify those impact categories which are decisive for the comparison, namely fossil fuel use, CO_2 emissions, use of raw phosphate, and N and P emissions to water. Finally, the urine separation scenario is assessed as preferable to the conventional system, with a high sensitivity to the subjective weighting procedure and to several assumptions concerning the surrounding technical systems.

Schneidmadl et al 2000

This LCA study evaluates different options for wastewater management and is associated with a pilot project testing new sanitation concepts in Freiburg, Germany ("Vauban") (Schneidmadl et al., 2000). The conventional system of wastewater and stormwater treatment is compared to several separation systems, including blackwater digestion, local infiltration of stormwater and the treatment of greywater in a sand filter with partial reuse as toilet flush water. For water-related emissions, the separation systems can substantially reduce the load of nutrients and COD to surface waters in comparison to the conventional system, especially due to the prevention of combined sewage overflow events. Similarly, the reduction of heavy metal emissions into surface waters leads to a lower aquatic ecotoxicity potential of the alternative systems. Concerning the energetic resources and related indicators such as the global warming potential, their advantages are less pronounced. In fact, some separation systems exhibit an even higher demand of fossil fuels and related CO_2 emissions, although the substitution of mineral fertilizer is taken into account. The construction of the infrastructure needs more resources for the separation systems, however the authors point out that the energy-related indicators are only of minor relevance for the comparison after normalisation. Consequently, the construction phase is found to be less important for the overall environmental performance of a sanitation system.

Jeppsson and Hellström 2002

This Swedish study compares two scenarios for urban wastewater systems using material flow analysis in combination with evaluation methods based on Life Cycle Assessment (Jeppsson and Hellström, 2002). The conventional centralised wastewater system includes activated sludge treatment with denitrification and chemical P elimination. Additionally, phosphorus is recovered from sewage sludge via the KREPRO process and recycled to agriculture. The second scenario is a separation system with urine separation, blackwater digestion, and greywater treatment in an activated sludge plant without enhanced phosphorus removal. The material flows of these two systems are simulated with ORWARE software and evaluated with LCA indicators. Due to the high recycling rate of phosphorus through the KREPRO process (80%), the conventional system offers a higher amount of recycled phosphorus and less cadmium input to agricultural soil than the separation system. The separation system features higher recycling of nitrogen and potassium and decreases the fresh water consumption. Both systems are net producers of energy due to the assumed use of heat pumps for the recovery of thermal energy from the wastewater. The authors conclude that neither of the investigated systems can be described as an absolute best solution based on the priority environmental criteria developed in this study. The outcome depends on local conditions and political priorities of decision-makers concerning relevant impact categories.

Mühleck et al 2003

Mühleck and co-authors describe a decision support system (DSS) which is developed for the assessment of environmental impacts of different wastewater management options in urban areas (Mühleck et al., 2003). The DSS is based on a material flow analysis (MFA) of the technical system and its evaluation with a set of environmental indicators. The authors want to establish a methodology for the assessment of wastewater management strategies on a regional level in contrast to site-specific studies. All relevant components of the Berlin drinking water and wastewater system are included in the MFA. The set of indicators is a mixture of typical LCA-derived indicators, water management-specific figures (e.g. water consumption per capita and year), and more legislative-oriented parameters (e.g. emissions to water are weighted in "damage units" of the German Wastewater Charges Act). The comparison of the conventional system with separation scenarios (blackwater separation with vacuum toilets or urine separation, both with nutrient recycling to agriculture) results in a significant reduction of nutrient emissions into receiving surface waters. The authors also identify a substantial reduction in heavy metal input to farmland in comparison to the spreading of sewage sludge in the conventional scenario. The transport of digested blackwater to farmland is seen as a possible hotspot with increased greenhouse gas emissions due to the high volume and low nutrient concentration of the organic fertilizers. However, if the substitution of mineral fertilizer products is accounted for, the separation scenarios show comparable or less emissions of greenhouse gases than the conventional system.

Hospido et al 2004

This LCA study is designed to evaluate the environmental impacts corresponding to a municipal wastewater treatment plant (90,000 inhabitant equivalents) with primary and secondary treatment (no denitrification or chemical P elimination) (Hospido et al., 2004). The inventory is based on real data from a two year measurement campaign characterizing all input and output flows of the WWTP. The categories of eutrophication (WWTP effluent) and terrestrial ecotoxicity (sludge application in agriculture) are identified as decisive for the environmental performance. An enhanced nitrogen removal via denitrification results in a substantial reduction of the environmental impacts. Heavy metals are responsible for the high terrestrial ecotoxicity (mainly Cr, Hg, Zn), which may lead to alternative treatment strategies for WWTP sludge in the future.

Lundie et al 2004

This is a prospective LCA study of water supply and wastewater collection and treatment in the greater Sydney area for the year 2021 (Lundie et al., 2004). The complete metropolitan water system including water filtration plants, distribution, sewer system, wastewater treatment plants, and sludge treatment and disposal is modelled with LCI data from site-specific measurements. The environmental performance of the actual system is extrapolated into the future and compared to several scenarios including water demand management, increased energy efficiency, desalination of seawater, and upgrading of major WWTPs. The impact assessment reveals that demand management and energy efficiency measures result in an improvement in all impact categories, while desalination and WWTP upgrade show environmental trade-offs. However, the missing normalisation and weighting of the results prevents a comprehensive evaluation of the different scenarios. A combination of ecological measures in a new suburb ("greenfield scenario") reveals environmental benefits in all indicator categories on a per household basis. In general, LCA is found to be a useful tool for the examination of alternative future scenarios for strategic planning.

Lassaux et al 2007

Lassaux and co-workers conduct an LCA for the water cycle in the Walloon Region of Belgium (Lassaux et al., 2007). They include the water catchment, water treatment and supply, the sewer system, the wastewater treatment plants including sludge treatment, and water discharge without any treatment. Based on the data of more than 100 wastewater treatment plants, different scenarios are calculated which reflect an increasing connection rate to the public sewer and consequently less discharge of untreated wastewater. The inventory includes construction and operation of all facilities, and it is evaluated with Eco-Indicator 99 and CML 2001. Nitrogen is excluded from the impact assessment in the category of eutrophication. The authors conclude that an increasing rate of connection to a public sewer and wastewater treatment plants decreases the global environmental burden of the water cycle. Although wastewater treatment and sewer construction consume considerable amounts of energy and chemicals, this disadvantage is far outweighed by the reduced discharge of P and COD to surface waters due to increased wastewater treatment.

Benetto et al 2009

The authors describe a comparative LCA case study for an office building (40 persons) in Luxembourg (Benetto et al., 2009). They compare conventional wastewater treatment in a centralized activated sludge plant with a scenario of ecological sanitation, including

urine separation, faeces composting, and greywater treatment in a reed bed. The secondary functions of fertilizer substitution are accounted in three different ways: “cut off”, i.e. no allocation of environmental burdens of disposal (transport and application), “waste”, i.e. urine and faeces are considered as waste that has to be disposed, and “system expansion” where the substitution of mineral fertilizer is accounted for. The authors emphasize that the system expansion approach is discussed controversially in the LCA community. Impact assessment is described with several indicators on the midpoint and endpoint level. Their endpoint results show that separation systems decrease ecosystem quality damage (particularly due to decreased emissions of Al from precipitation, Zn, and Cu), whereas effects on climate change, human health and resource demand are worse than for the conventional system. On the midpoint level, the authors state benefits in eutrophication and drawbacks in acidification for the separation system. Transport distance of organic fertilizers and the allocation of the secondary functions are identified as decisive points in sensitivity analysis.

Table 1: Overview of previous LCA case studies for wastewater systems

Source		Scale		Faeces treatment		Urine	Grey-water	System including		Substi-tution		Impact assessment				Remarks
	Country	*Inhabitant equivalents*	*CAS system*	*digestion*	*composting*	*separation*	*Alternative treatment*	*Infrastructure*	*Stormwater*	*Energy*	*Fertilizer*	*Resources*	*Water and air emissions*	*Soil emissions*	*Normalisation*	
Bengtsson et al., 1997	S	200 + 2700	X		X	X		(X)			X	X	X		(X)	**Two case studies**
Roeleveld et al., 1997	NL	100000	X					X				X	X		X	
Tillman et al., 1998	S	900 + 12600	X	X		X	X	X		X	X	X	X			**Heat recovery from WW in CAS system**
Schneidmadl et al., 2000	D	40	X	X			X	X	X	X	X	X	X		X	**Greywater reuse**
Jeppsson and Hellström, 2002	S	15000	X	X		X			X			X	X	(X)		**CAS with KREPRO (P recycling)**
Mühleck et al., 2003	D	4.1 Mio	X	X		X			X	(X)	X	X	X	X	(X)	**Berlin water supply and wastewater treatment**
Hospido et al., 2004	E	90000	X						X			X	X	X	X	
Lundie et al., 2004	AU	~ 4 Mio	X					X		X	X	X	X	X		**Detailed model of Sydney water system**
Lassaux et al., 2007	B	Walloon region	X					X	(X)		X	X	X		X	**Complete water cycle**
Benetto et al., 2009	L	40	X		X	X	X		X		(X)	X	X	(X)	X	
This study	**D**	**5000**	**X**	**X**	**X**	**X**	**X**	**X**		**X**	**X**	**X**	**X**	**X**	**X**	**Greywater reuse via MBR treatment**

2.3 Approach of this study

The approach of this study targeting a comparison of environmental impacts of different wastewater management systems is developed in consideration of the outcomes of previous studies. It attempts to combine all relevant issues addressed in previous research (Table 1), especially concerning the applied methodology of LCA and the investigated types of separation systems. In detail, the following aspects are implemented (Figure 5):

- The study follows the methodology of LCA as described in ISO 14040/44. All relevant steps shall be considered (excluding critical review).
- As a prospective LCA for strategic planning, a hypothetical case study is set up in which different options for the management of municipal wastewater (excluding stormwater) and biowaste are compared.
- The scale of the case study is set to 5000 inhabitants, thus reflecting an urban environment in which an implementation of separation systems could be possible within a reasonable timeframe.
- Different options for the layout of separation systems (2-flow, 3-flow, non-potable reuse) are included, combining available technology for each partial flow of wastewater.
- The system boundaries include operation and construction of wastewater systems, associated background processes, and secondary functions.
- Specific modular substance flow models for each relevant process of the core system of wastewater management are set up with available LCI data compiled from pilot projects and literature, using the LCA software UMBERTO® (IFU and IFEU, 2005).

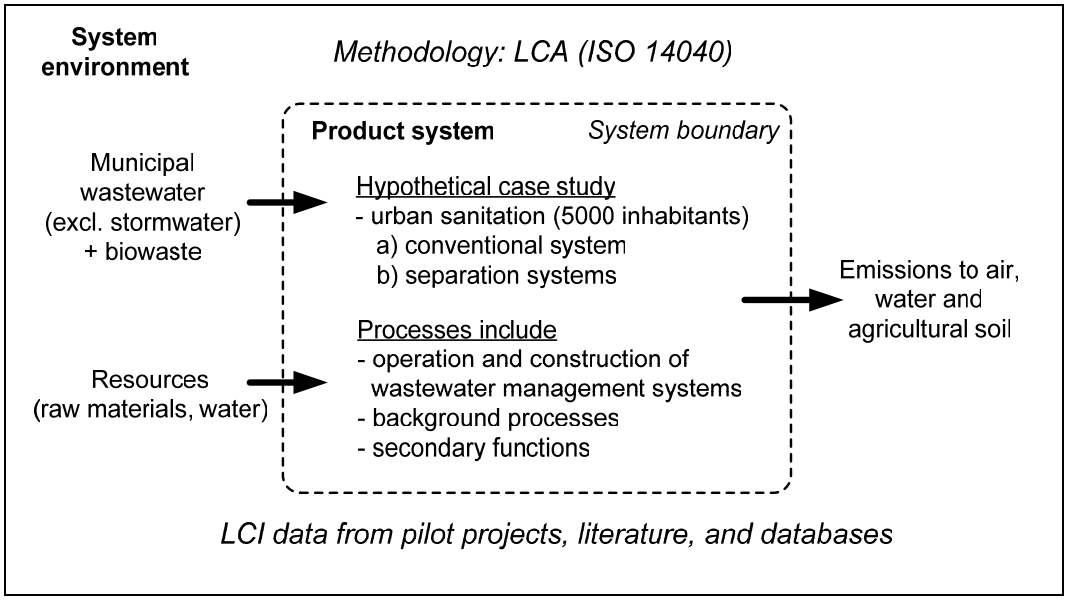

Figure 5: Approach of this study

3 Definition of goal and scope

3.1 Goal and target group

In this study, different systems for the management of urban wastewater are analysed in order to compare the systems in terms of selected impacts on the environment. The goal is to identify advantages and disadvantages of the investigated systems with special regard to possible benefits of source-separation technologies compared to conventional wastewater systems. On the other hand, potential ecological hotspots of separation systems shall be revealed to specify those parts of the system which need further improvement. Finally, key parameters with high relevance for the ecological assessment shall be identified, so that future LCA studies in this field may use a simplified approach for setting up the Life Cycle Inventory.

No existing marketable products are compared, but integrated concepts of urban wastewater management. The results of this work are primarily intended for experts from research and sanitary engineers working in the fields of urban wastewater management and environmental assessment of water systems. The results may also be interesting for professionals in administrative or political functions related to decision-making in the wastewater sector.

3.2 Function and functional unit

The core or primary function of the sanitation systems under investigation is the transport and disposal of all fractions of urban household wastewater. To reflect possible advantages of an integrated approach of managing municipal waste, the disposal of solid biowaste is also defined as a primary function. Additionally, the supply of drinking water is included as a primary function to reflect the effects of varying freshwater demand between the scenarios. In summary, the primary functions include the following services:

Primary functions

- *supply of drinking water for urban households*
- *transport and disposal of human urine and faeces from urban households*
- *drainage and treatment of domestic urban wastewater charged with substances from kitchen, personal hygiene, and washing machines (greywater)*
- *collection and disposal of solid domestic biowaste from kitchen and garden*

This study explicitly excludes all other fractions of wastewater which may arise in a settlement, i.e. stormwater which is often collected together with domestic sewage or wastewater from industrial or agricultural activity. The scope of the study is purposely limited to domestic wastewater from households, because the influence of other wastewater fractions on the wastewater system is heavily depending on local conditions (e.g. combined or separate collection of rainwater, amount of rainfall, type and size of industrial or agricultural activity). This limitation keeps the size of the study manageable without compromising the general validity of its results for urban wastewater systems.

The material and energy flows associated with the above mentioned primary functions, respectively the LCA results, are related to the provision of these services per person and year. Consequently, the functional unit is defined as:

Functional unit:	*Provision of the primary functions per person and year* → *Unit: $(pe*a)^{-1}$*

Besides their primary functions, separation systems additionally provide secondary functions, i.e. the supply of organic fertilizers and electric energy. For proper system comparison, these secondary functions are taken into account by the method of system expansion (see chapter 3.4).

Although the functional unit is related to the provision of sanitation services per person, the overall size of the settlement under investigation is important as well. For certain facilities such as collection systems or treatment plants, a minimum scale is required for reliable and cost-efficient operation. Additionally, the settlement size affects other boundary conditions (e.g. transport distances). It is assumed that the sanitation systems shall be installed in an urban settlement with 5000 inhabitants with low to medium population density (~ 40 inhabitants per ha). This setting can be seen as exemplary for a small town or a suburban city district with a mixture of single houses and apartment blocks. Consequently, the results of this study are not directly applicable for smaller or rural settlements (< 500 inhabitants), because the technical implementation and process parameters would be different for small-scale systems.

3.3 Reference input flows

The amount and elemental composition of the reference input flows of drinking water, wastewater constituents and biowaste are defined based on data collected in an extensive review of relevant literature (cf. annex 12.1). Thus, average values are generated for each input flow which are supposed to represent a typical composition of domestic wastewater in Germany. Recently, a similar compilation of literature data was

published to determine key figures for the composition of household wastewater flows in Germany (Oldenburg et al., 2008).

Daily mass flows of urine, faeces (+ toilet paper), and greywater per person are defined as 1.5, 0.14, and 80 kg/(pe*d), respectively (Table 2). The average daily excretion of urine and faeces of an average adult is relatively well-known from medical studies (Ciba-Geigy, 1977). Greywater volume and quality can vary considerably depending on its origin and user behaviour (Li et al., 2009). This study assumes an average amount of 80 L of greywater per person and day, which is on the lower range of published values and reflects a general trend of decreasing water consumption in Germany, mostly due to water-saving household appliances. The total domestic water demand in Germany (including toilet flush water) ranges from 90 L to 143 L per person and day, with an average of 126 L/(pe*d) in 2004 (DESTATIS, 2006).

The average quantity of kitchen and garden biowaste is characterized by large variations depending e.g. on user behaviour, the structure of the urban area and the annual seasons. The amount of garden biowaste which is assumed in this study represents the average potential of organic garden waste in Germany (0.3 kg/pe*d). In addition, there is an average potential of 0.2 kg/(pe*d) of loppings from municipal greens (Wintzer et al., 1996). The annual range of the mass flow of mixed urban biowaste is about factor 2, the range of the volume flow about factor 3 (Fricke, 1990). It is emphasized that the potential amount of domestic garden biowaste can be higher than assumed in this study.

Table 2: Mass flow of reference input flows

Input flow	Mass flow	
	*kg/(pe*a)*	*kg/(pe*d)*
Urine	547.5	1.5
Faeces* + toilet paper	51.1	0.14
Toilet flush water**	8760 – 13140 1898 – 8760	24 – 36 (conventional) 5.2 – 24 (separation)
Greywater	29200	80
Kitchen biowaste*	58.4	0.16
Garden biowaste*	109.5	0.3

Sources: annex 12.1
** wet mass*
*** depending on type of toilet used in scenario*

The defined elemental composition of the reference input flows in terms of organic matter, nutrients, salts, and heavy metals is assumed to represent average values for Germany (Table 3 and Table 4). It should be pointed out that the amount of nutrients in human excreta is heavily depending on the nutritional diet (Ciba-Geigy, 1977). The composition of drinking water is estimated (contributing to both toilet wastewater via flush water and greywater loads) and represents typical concentrations after contact with pipe materials (Schulz et al., 2008). The dissolution of copper and zinc from pipe materials is responsible for the elevated concentrations of these heavy metals in drinking water (here: Cu = 0.16 mg/L, Zn = 0.37 mg/L). Thus, drinking water contributes a major fraction of the total loads of these metals in the input flows. In scenarios with wastewater reuse for toilet flushing, the composition of flush water is calculated according to effluent concentrations of the respective wastewater treatment process.

Table 3: Average composition of faeces, urine, and greywater

Flow		Urine	Faeces (+ toilet paper)	Greywater*
Quantity	*kg/(pe*d)*	1.50	0.14	80
Main constituents and nutrients				
Dry matter	*g/(pe*d)*	60	45	120
Organic dry matter	*g/(pe*d)*	45	42	
COD	*g/(pe*d)*	15	35	60
TOC	*g/(pe*d)*	7	21	18
N total	*g/(pe*d)*	10	1.5	1.3
P total	*g/(pe*d)*	1	0.5	0.5
K	*g/(pe*d)*	2.6	0.55	2
Na	*g/(pe*d)*	3.5	0.15	6
Ca	*g/(pe*d)*	0.21	1	14
Mg	*g/(pe*d)*	0.12	0.2	3
Cl	*g/(pe*d)*	4.8	0.06	7
S total	*g/(pe*d)*	0.8	0.2	7.5
Heavy metals				
Cd	*mg/(pe*d)*	0.0002	0.02	0.2
Cr	*mg/(pe*d)*	0.01	0.02	3
Cu	*mg/(pe*d)*	0.05	1.5	20
Hg	*mg/(pe*d)*	0.0004	0.02	0.02
Ni	*mg/(pe*d)*	0.04	0.2	2
Pb	*mg/(pe*d)*	0.01	0.02	3
Zn	*mg/(pe*d)*	0.25	10	46

** including loads from drinking water (80 L, for composition see Table 4)*
Sources: annex 12.1

Table 4: Average composition of kitchen and garden biowaste and drinking water

Flow		Kitchen biowaste	Garden biowaste	Drinking water*	
Quantity	*kg/(pe*d)*	0.16	0.30	(amount of flush water depends on scenario)	
Main constituents and nutrients					
Dry matter	*% wet mass*	50	41	520	mg/L
Organic dry matter	*g/(pe*d)*	36	87		
TOC	*g/(pe*d)*	13	45.5	1.10	mg/L
N total	*g/(pe*d)*	0.9	1.4	1.00	mg/L
P total	*g/(pe*d)*	0.2	0.6	0.08	mg/L
K	*g/(pe*d)*	0.6	13.6	7.50	mg/L
Na	*g/(pe*d)*	1.2	0.2	36.00	mg/L
Ca	*g/(pe*d)*	1	4.1	103.00	mg/L
Mg	*g/(pe*d)*	0.22	0.6	10.00	mg/L
Cl	*g/(pe*d)*	3	0.04	18.00	mg/L
S total	*g/(pe*d)*	0.1	0.06	40.50	mg/L
Metals					
Cd	*mg/(pe*d)*	0.01	0.05	0.0005	mg/L
Cr	*mg/(pe*d)*	0.5	0.6	0.005	mg/L
Cu	*mg/(pe*d)*	1	2.3	0.16**	mg/L
Hg	*mg/(pe*d)*	0.01	0.02	0.0002	mg/L
Ni	*mg/(pe*d)*	0.2	0.5	0.005	mg/L
Pb	*mg/(pe*d)*	0.6	0.6	0.005	mg/L
Zn	*mg/(pe*d)*	7.3	13.5	0.37**	mg/L

Sources: annex 12.1
** used as flush water (except wastewater reuse scenarios)*
*** elevated due to contact with pipe materials*

Assumed total amounts of organic matter, nitrogen and phosphorus in wastewater flows are comparable to population equivalents for municipal wastewater in Germany (Table 5). Population equivalents of DWA are empirical values used for the dimensioning of wastewater treatment plants and represent a maximum daily load which is not exceeded in 85% of days (ATV, 2000). They offer a reasonable basis for estimating the validity of assumed wastewater composition in this study.

Table 5: Allocation of organic matter, nitrogen and phosphorus in wastewater flows compared to ATV population equivalents

		Urine	Faeces	Greywater	SUM	Population equivalent* of ATV A131
COD	*g/(pe*d)*	15	35	60	**110**	**120**
N	*g/(pe*d)*	10	1.5	1.3	**12.8**	**11**
P	*g/(pe*d)*	1	0.5	0.5	**2.0**	**1.8**

** daily load per person that is not exceeded on 85% of days (ATV, 2000)*

3.4 System expansion

The primary function of the investigated sanitation systems is defined as the collection and safe disposal of wastewater and biowaste from households. Besides the primary function, sanitation systems can provide secondary functions of interest, namely the supply of secondary fertilizers and energy. Secondary fertilizers can be in the form of sewage sludge, compost, urine, faeces and related products (e.g. sludge from faeces digestion).

For a comprehensive comparison of all investigated systems, these secondary functions have to be included in the environmental assessment to avoid an allocation problem. A previous LCA study comparing urine separation to conventional wastewater treatment has revealed a significant influence of secondary functions of fertilizer substitution on the results of the impact assessment (Lundin et al., 2000).

A suitable procedure to consider secondary functions in LCA is the system expansion approach which is used in many LCA case studies. There are two different ways to implement system expansion in an LCA study: One approach broadens the system boundaries and introduces a new function or product to make the two systems being compared equal in scope (Figure 6). Another approach subtracts the environmental burdens of an alternative way of supplying the secondary function from the respective system ("avoided burden") (Curran, 2007).

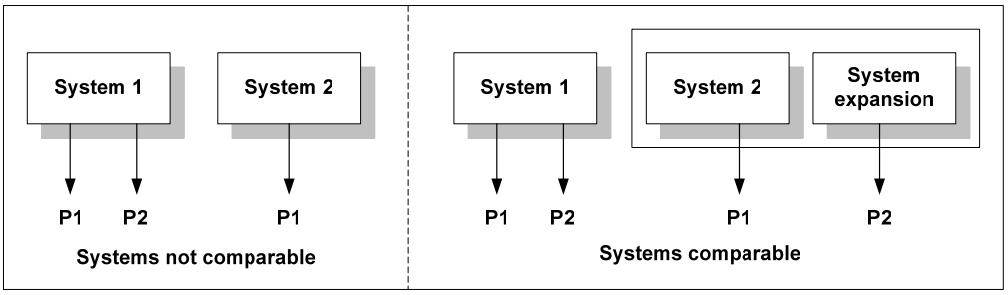

Figure 6 Principle of system expansion by broadening the system boundaries

In this study, the relevant scenarios are expanded with additional processes which deliver equivalent products, i.e. mineral fertilizer and electric energy (Table 6). Nitrogen, phosphorus, and potassium are taken into account as relevant macronutrients for agriculture. Regarding the potential of secondary fertilizers to substitute industrially produced mineral fertilizer, the nutrient availability of sewage sludge, urine, faeces and related products in relation to that of mineral fertilizer is decisive for calculating the respective amounts of mineral fertilizer that can be effectively substituted (= fertilizing equivalents). With the system expansion, all scenarios in this study supply exactly the same amount of fertilizing equivalents and electric energy.

Table 6: Secondary functions delivered by sanitation systems and their respective equivalent products in system expansion

Secondary functions of sanitation systems	Equivalent products for system expansion
Supply of macronutrients N, P, K with organic fertilizers from: ▪ Sewage sludge ▪ Urine ▪ Composted faeces or biowaste ▪ Residual of faeces digestion	Industrially produced mineral fertilizers
Supply of electric energy from combustion of: ▪ sewage gas from sludge digestion ▪ biogas from digestion of faeces and biowaste	Electric energy from the grid

Note: thermal energy and fertilizing value of organic carbon are excluded in this study

Exclusions

Two possible secondary functions have explicitly been excluded from system expansion: a) thermal energy (= heat) and b) organic carbon content of secondary fertilizers. For both products, the effective benefit is difficult to quantify in terms of equivalent products, which led to their exclusion in this assessment.

For thermal energy, the utility value of a possible surplus of heat heavily depends on the heat demand near the place of origin. Losses during heat transmission and transport can be substantial and depend on local conditions. However, thermal energy can be utilized within the system, e.g. waste heat from biogas combustion is used for the hygienisation process of digester substrate.

The beneficial effect of the organic carbon content of secondary fertilizers on soil fertility and crop growth cannot be precisely quantified. Organic fertilizers are mainly applied as supplementary fertilizers for soil conditioning. Thus, it is not clear whether the organic content of secondary fertilizers will in fact substitute other organic fertilizers (e.g. peat or bark mulch). It should be pointed out that this exclusion may lead to an underestimation of the beneficial secondary functions of organic fertilizers, especially in areas where the soil is depleted in organics.

3.5 Description of the investigated sanitation scenarios

The sanitation scenarios which are investigated in this study represent different options for collection and treatment of the various flows of household wastewater and biowaste within a settlement of 5000 inhabitants. The collection and treatment of stormwater as well as wastewater from industrial or agricultural activities is excluded in this study.

Reference scenarios

The conventional system with combined collection of all household wastewater flows in one sewer system and its treatment in a conventional activated sludge (CAS) plant is taken as a reference scenario. More precisely, three configurations of the conventional system are taken into account, representing different levels of treatment technology and options for sludge handling:

- **Scenario R**: The baseline scenario represents conventional wastewater treatment with extended removal of nutrients N and P and optimized energy demand (anaerobic sludge digestion and use of sewage gas). Stabilised sludge is incinerated and disposed in a landfill. Many wastewater treatment plants in Germany are operated with extended nutrient removal to minimize nutrient loads into surface waters (DWA, 2005), even though it is not explicitly required by legal standards for this dimension (5000 population equivalents = size range 2, in: AbwV, 2004).
- **Scenario R_{min}**: In this scenario, wastewater is only treated to comply with current legal discharge standards for a wastewater treatment plant of this dimension, i.e. no extended nutrient removal is required. This scenario reflects the minimum requirements in terms of effluent quality. Sewage sludge is stabilised by extended aeration and incinerated without the use of sewage gas, i.e. the plant is not energetically optimized.
- **Scenario R_{agri}**: The third reference scenario applies advanced wastewater treatment similar to scenario R, but the digester residual from sludge stabilisation is directly applied in agriculture. This scenario reflects the maximum nutrient recovery potential in a conventional system.

In all reference scenarios, household biowaste is separately collected before it is transported to incineration or composting. While incineration is a traditional disposal route for all types of municipal solid waste, composting represents a specific option for biowaste handling to recover a valuable product (compost). It is assumed here that 50% of the garden biowaste and 20% of the kitchen biowaste is incinerated, while the remaining biowaste is converted into compost.

Separation scenarios

In these scenarios, the different wastewater flows are collected and treated separately. Thus, an enhanced recycling of valuable resources such as nutrients and freshwater is possible. If wastewater flows are separately collected, different treatment processes are available for the respective flows of urine, faeces, and greywater. This study investigates three different options for collection and treatment of urine and faeces (= toilet wastewater):

- **Scenario group V**: combined collection of urine and faeces with a vacuum system and co-digestion with biowaste, application of digester residual in agriculture
- **Scenario group SV**: separate collection of undiluted urine and direct application as fertilizer, vacuum collection of faeces and co-digestion with biowaste, application of digester residual in agriculture
- **Scenario group SC**: separate collection of undiluted urine and direct application as fertilizer, gravity collection of faeces and composting with biowaste, application of compost in agriculture

Furthermore, there are different options for the treatment of greywater (Li et al., 2009), which is collected and treated separately in all separation scenarios. This study includes three options for greywater treatment:

- **1st option**: treatment in a conventional activated sludge plant which is operated as a sequencing batch reactor (= SBR)
- **2nd option**: near-natural treatment in a planted soil filter (= constructed wetlands)
- **3rd option**: treatment in a membrane bioreactor (= MBR) with subsequent non-potable reuse of a part of the effluent for toilet flushing

Each option for greywater treatment is combined with the three scenario groups for treatment of urine and faeces, resulting in a total of nine separation scenarios. While scenario group V represents a two-flow system (toilet wastewater and greywater), scenario groups SV and SC are three-flow systems (undiluted urine, faeces plus flush water, and greywater).

Including the three reference scenarios, this study investigates twelve different scenarios for wastewater treatment (Table 7). In the following chapter, each scenario is described in detail for a precise definition of the processes and flows within the systems which are relevant for this study.

Table 7: Overview of the investigated sanitation scenarios

Scenario	Urine	Faeces	Greywater	Biowaste*
R	Combined collection and treatment in CAS plant (extended nutrient removal, sludge digestion and incineration)			
R_{min}	Combined collection and treatment in CAS plant (only nitrification, aerobic sludge stabilisation and incineration)			Composting → fertilizer
R_{agri}	Combined collection and treatment in CAS plant (extended nutrient removal, sludge digestion and application in agriculture)			
V1			SBR	
V2	Vacuum drainage and digestion → energy + fertilizer		Soil filter	Digestion → energy and fertilizer
V3			MBR + reuse	
SV1			SBR	
SV2	Separate collection → fertilizer	Vacuum drainage and digestion → energy + fertilizer	Soil filter	Digestion → energy and fertilizer
SV3			MBR + reuse	
SC1			SBR	
SC2	Separate collection → fertilizer	Gravity drainage and composting → fertilizer	Soil filter	Composting → fertilizer
SC3			MBR + reuse	

** 50% of garden biowaste and 20% of kitchen biowaste is incinerated in each scenario*
CAS: conventional activated sludge
SBR: sequencing batch reactor
MBR: membrane bioreactor

3.5.1 Reference scenarios (R)

Scenario R

The reference scenario uses modern low-flush toilets (6L flush volume). The different household wastewater streams are collected in a combined sewer system and drained by gravity. Wastewater is treated in an activated sludge plant operating with extended nutrient removal, including the elimination of organic matter, nitrogen (conversion of dissolved nitrogen to N_2 via nitrification and denitrification) and phosphorus (chemical precipitation with ferric salts) (Figure 7). The excess sludge is stabilised separately by anaerobic digestion, dewatered and co-incinerated in a municipal waste incineration plant. Sludge liquor is recycled back to the influent of the plant. Sewage gas is combusted in a central heat and power (CHP) plant to produce electricity and heat. Collected biowaste is composted and applied in agriculture.

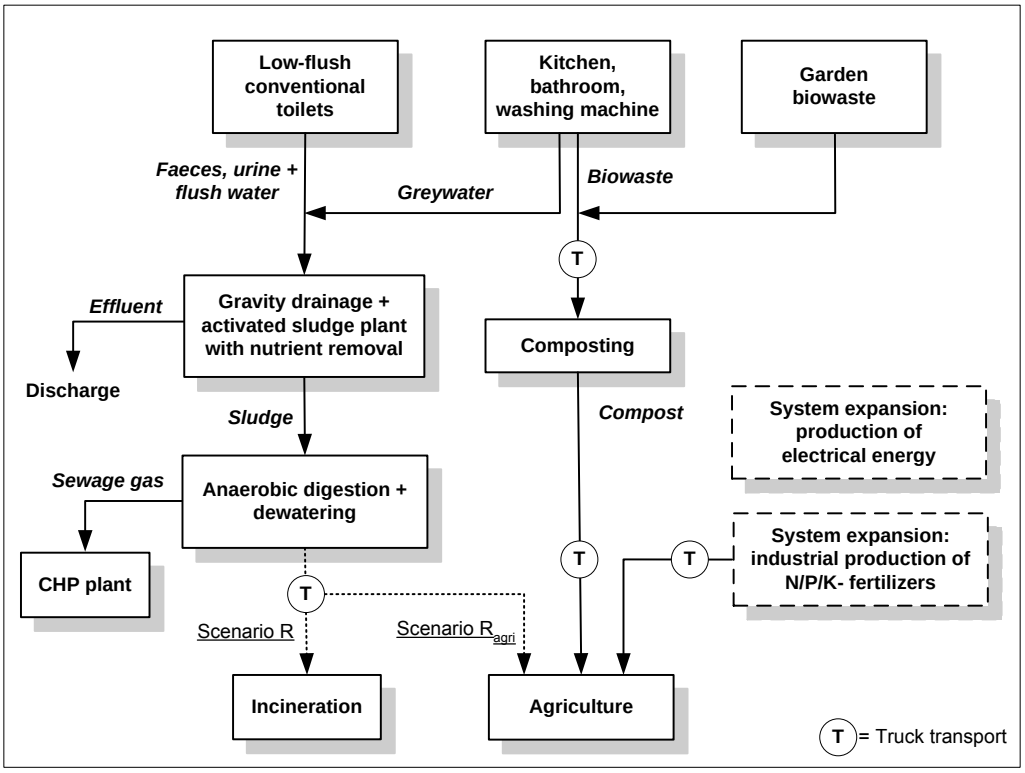

Figure 7: System setup of reference scenarios with advanced treatment (R and R_{agri})

Scenario R_{min}

This scenario features conventional toilets with normal flush volume (9L), a combined gravity sewer, and a conventional activated sludge plant for elimination of organics and ammonia (nitrification = conversion of NH_4 into NO_3). The sludge is aerobically stabilised by extended aeration, dewatered and co-incinerated in a municipal waste incineration plant (Figure 8). Sludge liquor is recycled back to the influent of the plant. Collected biowaste is composted and applied in agriculture.

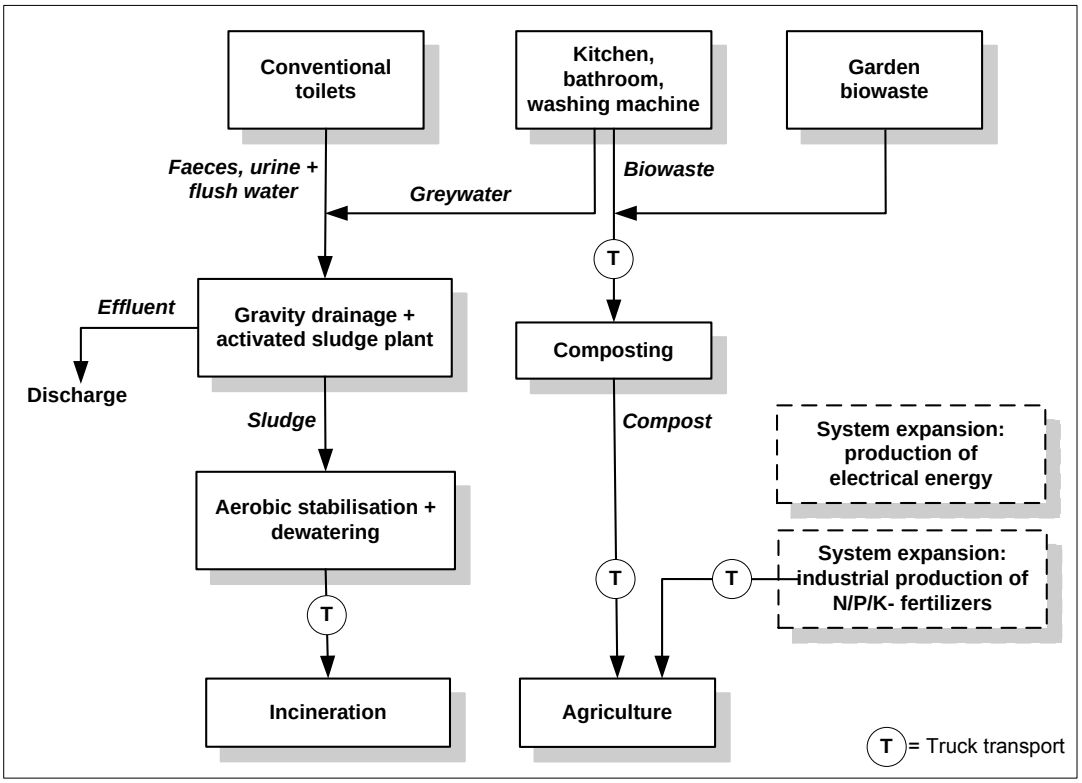

Figure 8: System setup of reference scenario with minimum treatment (R_{min})

Scenario R_{agri}

System setup of this scenario is exactly the same as in the baseline scenario R, except for the sludge disposal. In this scenario, stabilised sludge is dewatered and directly applied in agriculture as organic fertilizer (Figure 7). Thus, nutrient recycling is maximised in this scenario for a conventional sanitation system.

3.5.2 Separation scenarios with vacuum collection and digestion of urine and faeces (V)

In this group of separating scenarios, vacuum toilets are used to drain the mixture of faeces and urine with small amounts of flush water (0.7 – 1.2 L per flush). Mixed toilet wastewater is pumped from the vacuum station to a biogas plant, where it is digested together with household biowaste. Biogas is combusted in a CHP plant to generate electricity and heat. The residual digester sludge contains large amounts of valuable nutrients, but also a lot of water. The majority of the nutrients in this sludge derive from urine and are likely to be lost for recycling purposes with the filtrate if the residual sludge is dewatered and the filtrate is separated. Hence, it is assumed that this sludge is directly applied in agriculture without dewatering. This allows a high amount of nutrients to be recycled, but also creates large volumes of sludge which have to be transported to the point of application.

Scenario V1

Toilet wastewater is drained in a vacuum sewer and treated by anaerobic digestion in a biogas plant together with household biowaste. The residual sludge is directly applied in agriculture. Greywater is drained by gravity and treated in a conventional activated sludge process. This process is operated as a sequencing batch reactor (SBR) with partial denitrification and chemical P elimination (Figure 9). Excess sludge is aerobically stabilised, dewatered and incinerated.

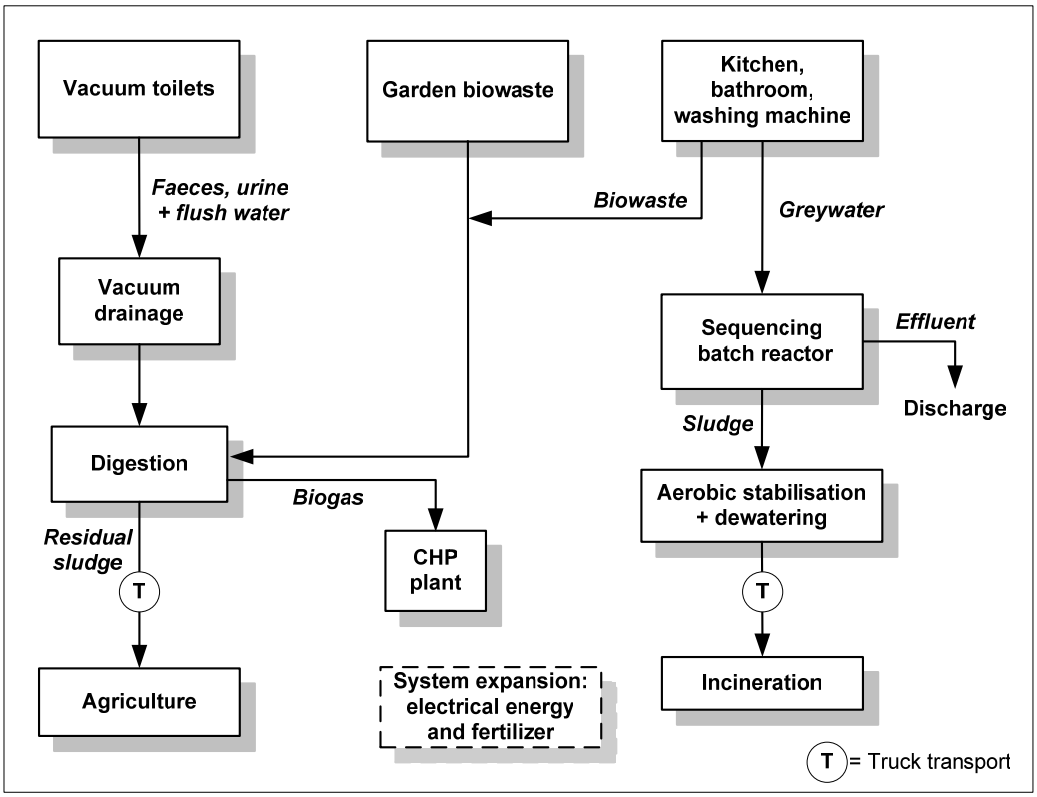

Figure 9: System setup of digestion scenario with SBR (V1)

Scenario V2

In this two-flow scenario, greywater is treated in a soil filter after extended settling in a sedimentation tank (Figure 10). The soil filter is planted with reed to prevent clogging of the filter and provide additional oxygen for biological processes in the filter bed. The reed is mowed once each year, and the plants are disposed in the biogas plant. Primary sludge from sedimentation is incinerated without dewatering.

Scenario V3

Here, greywater is treated in a membrane bioreactor to reach a high quality effluent. This effluent is partially reused for toilet flushing (Figure 11). Therefore, a separate pipe network for reused water has to be installed in the settlement. Excess sludge from the MBR is aerobically stabilised, dewatered and incinerated.

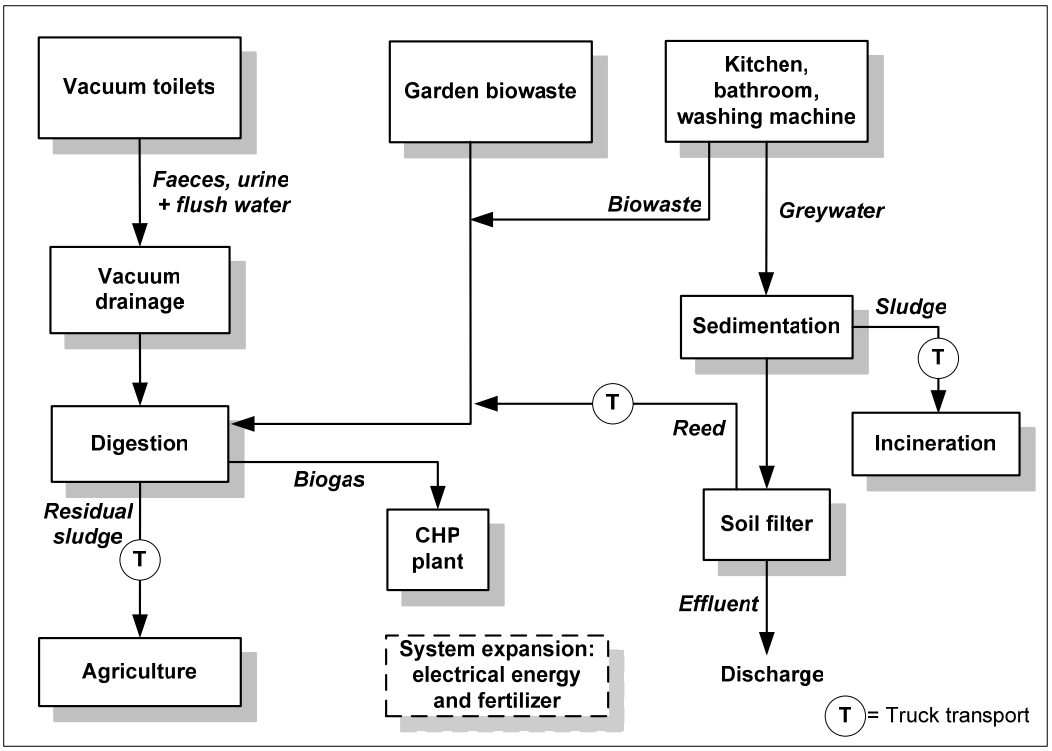

Figure 10: System setup of digestion scenario with soil filter (V2)

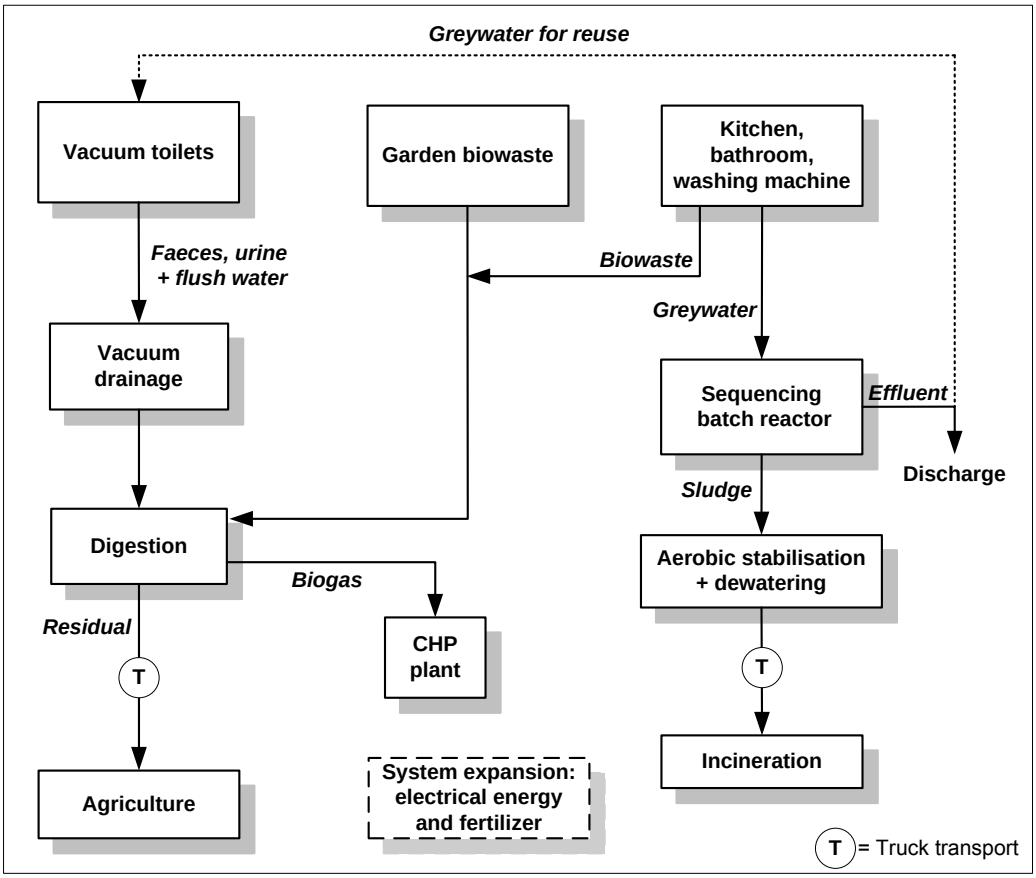

Figure 11: System setup of digestion scenario with MBR and reuse (V3)

3.5.3 Separation scenarios with urine separation, vacuum drainage and digestion of faeces (SV)

In this scenario group, vacuum separation toilets are applied. These toilets combine the urine separation technology (Roediger, 2007) with a vacuum system for the transport of faeces and flush waster. A special valve construction allows the separate collection of urine which is not diluted with flush water. Although this type of toilets is not yet available on the market, prototypes have been successfully tested in pilot projects (Peter-Fröhlich et al., 2007). Depending on user behaviour, it was found that 60-90% of total urine can be collected (Jönsson, 2001).

Undiluted urine is collected in separation toilets and drained by gravity to large storage tanks. After oxidative treatment by ozone for the removal of potentially harmful pharmaceuticals and other micro-pollutants, it is directly applied as liquid fertilizer in agriculture. Faeces and small amounts of flush water (0.7 – 1.2 L) are drained by a vacuum system and pumped to a biogas plant, where they are digested together with household biowaste. Generated biogas is combusted in a CHP plant. The residual digester sludge still contains valuable nutrients, and it is used as fertilizer in agriculture after dewatering and aerobic stabilisation (composting).

The toilet flush water separated as sludge liquor of the digester residual is now highly loaded with organics and nutrients. Therefore, it is decided to treat this concentrate separately in a biological reactor (activated sludge process) with high efficiencies in organics and nutrient removal. This is in contrast to scenario group SC (see 3.5.4), where the toilet flush water is separated from faeces before composting and treated together with greywater. Thus, the influent load for greywater treatment is considerably lower in SV scenarios than in SC scenarios.

Scenario SV1

In scenario SV1, greywater is treated in an activated sludge process operated as a sequencing batch reactor (Figure 12). The process includes nitrification, partial denitrification, and chemical P elimination. Excess sludge is aerobically stabilised, dewatered and incinerated.

Scenario SV2

Scenario SV2 includes greywater treatment in a planted soil filter after extended settling in a sedimentation tank (Figure 13).

Scenario SV3

In scenario SV3, greywater is treated in a membrane bioreactor with high effluent quality. The effluent is partially reused for toilet flushing (Figure 14).

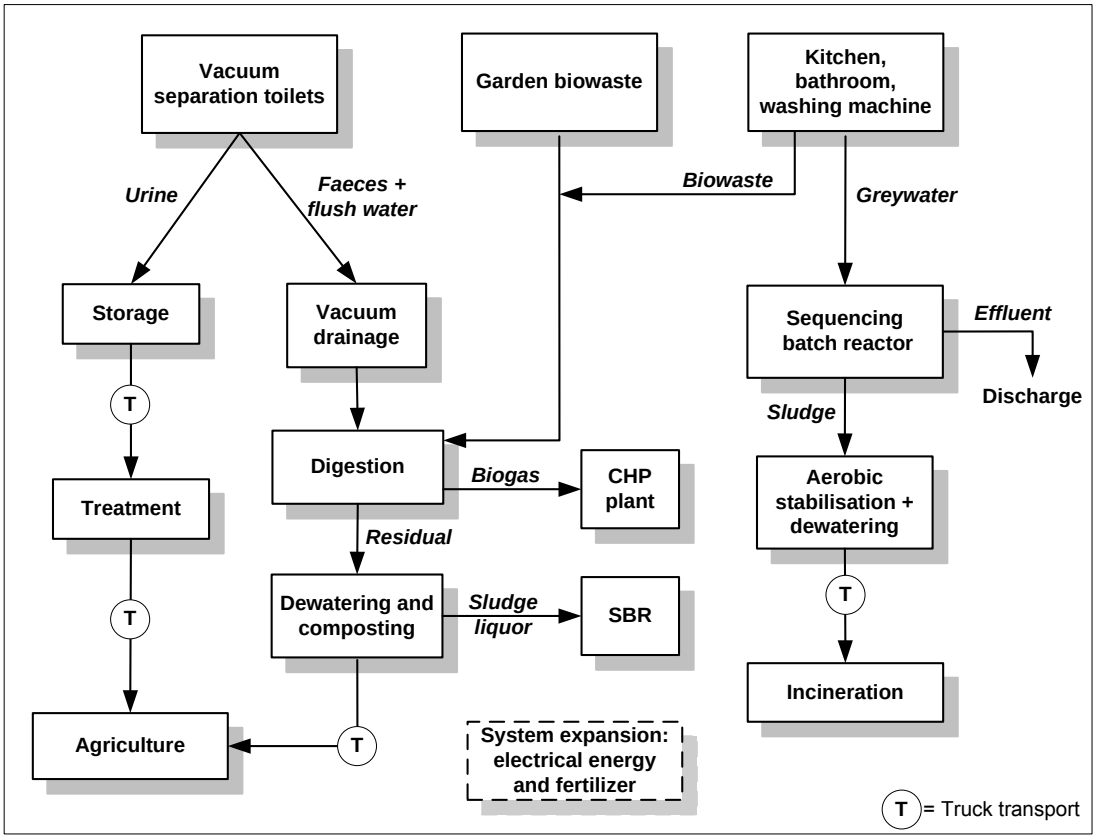

Figure 12: System setup for digestion scenario with urine separation and SBR (SV1)

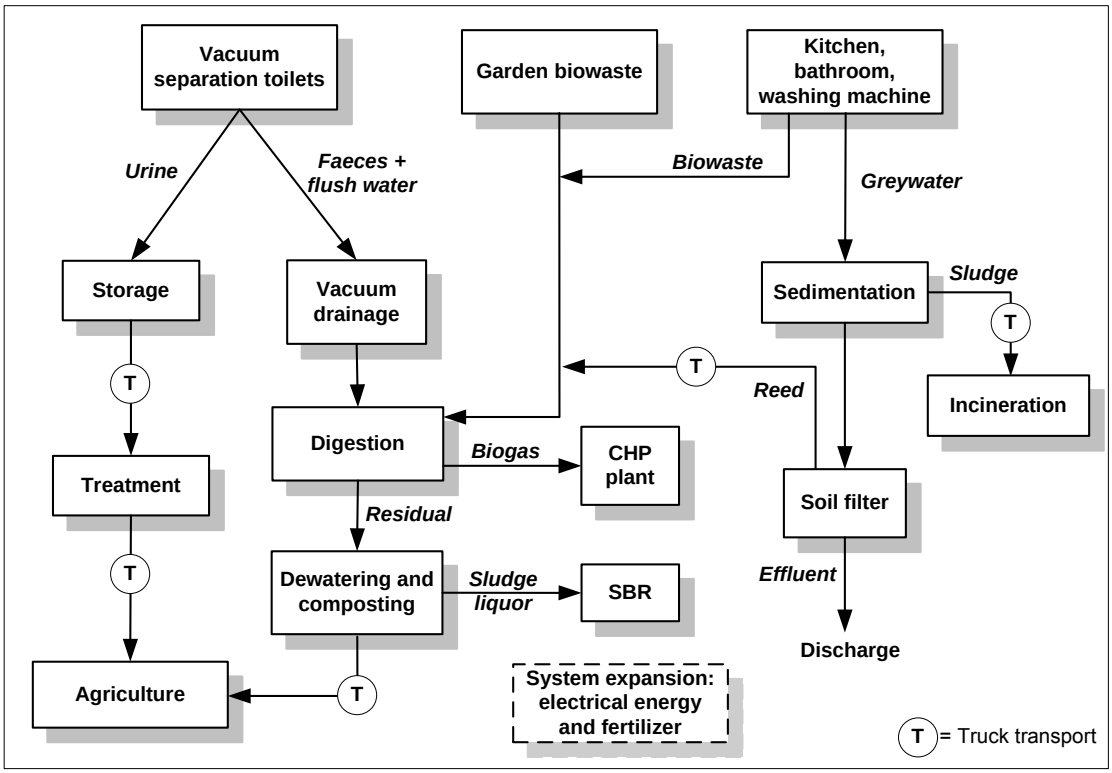

Figure 13: System setup of digestion scenario with urine separation and soil filter (SV2)

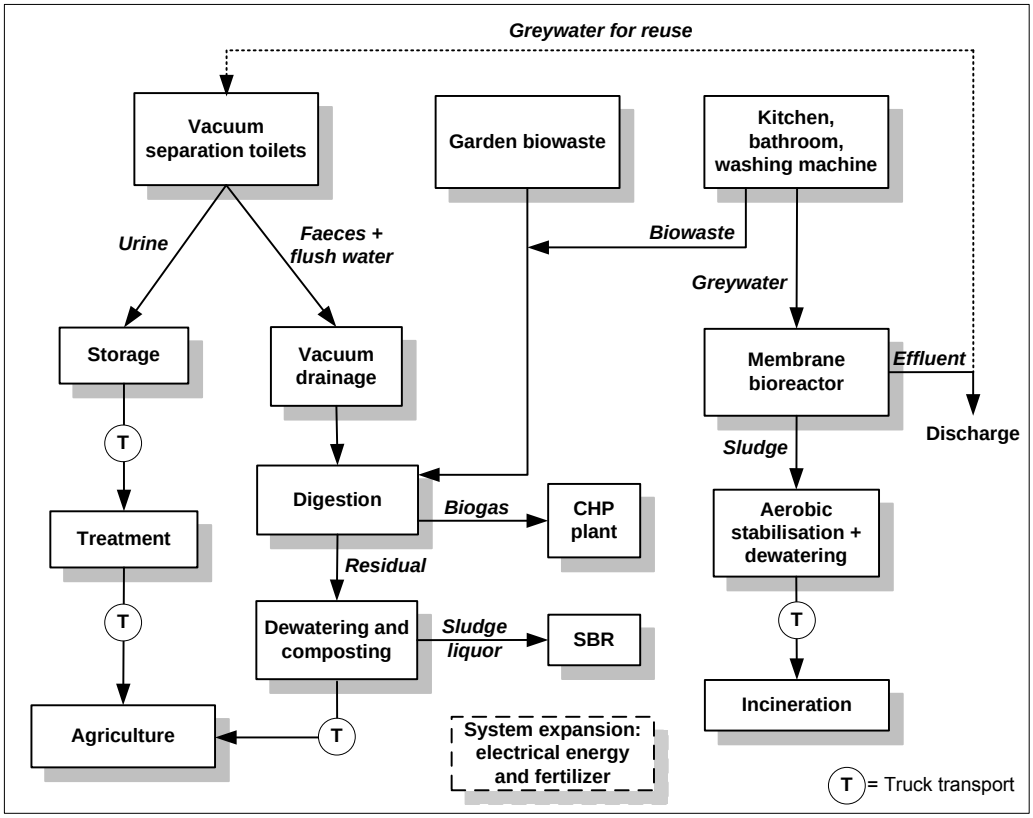

Figure 14: System setup for digestion scenario with urine separation, MBR and reuse (SV3)

3.5.4 Separation scenarios with urine separation and composting of faeces (SC)

In this scenario group, urine separation toilets are used to separately collect undiluted urine at the source. Undiluted urine is drained by gravity and collected in tanks, stored, and treated by ozonation for the elimination of micro-pollutants. Thereafter, it is transported to farms and directly applied as a multi-nutrient fertilizer in agriculture. Faeces are drained with flush water (6 L per flush) in a gravity system. After flush water has been mechanically separated from solids, faecal matter is stabilised in an aerobic composting process together with biowaste. Whereas compost is used as soil conditioner in agriculture, faeces filtrate with high volume and low nutrient loads is pumped to greywater treatment.

Scenario SC1

In the first scenario, greywater and faeces filtrate are treated in an activated sludge plant operated as a sequencing batch reactor with extended nutrient removal (Figure 15). Excess sludge is stabilised aerobically, dewatered and incinerated.

Scenario SC2

The second composting scenario includes a soil filter for the treatment of greywater and faeces filtrate (Figure 16). The process comprises of primary sedimentation, addition of faeces filtrate, and a planted soil filter which eliminates organics and a part of the nutrients. Reed from the soil filter is mowed once each year and added to the composting process. Sludge from greywater sedimentation stage is directly dewatered and incinerated. An additional stabilisation process for primary sludge is neglected here due to the small amounts of this sludge.

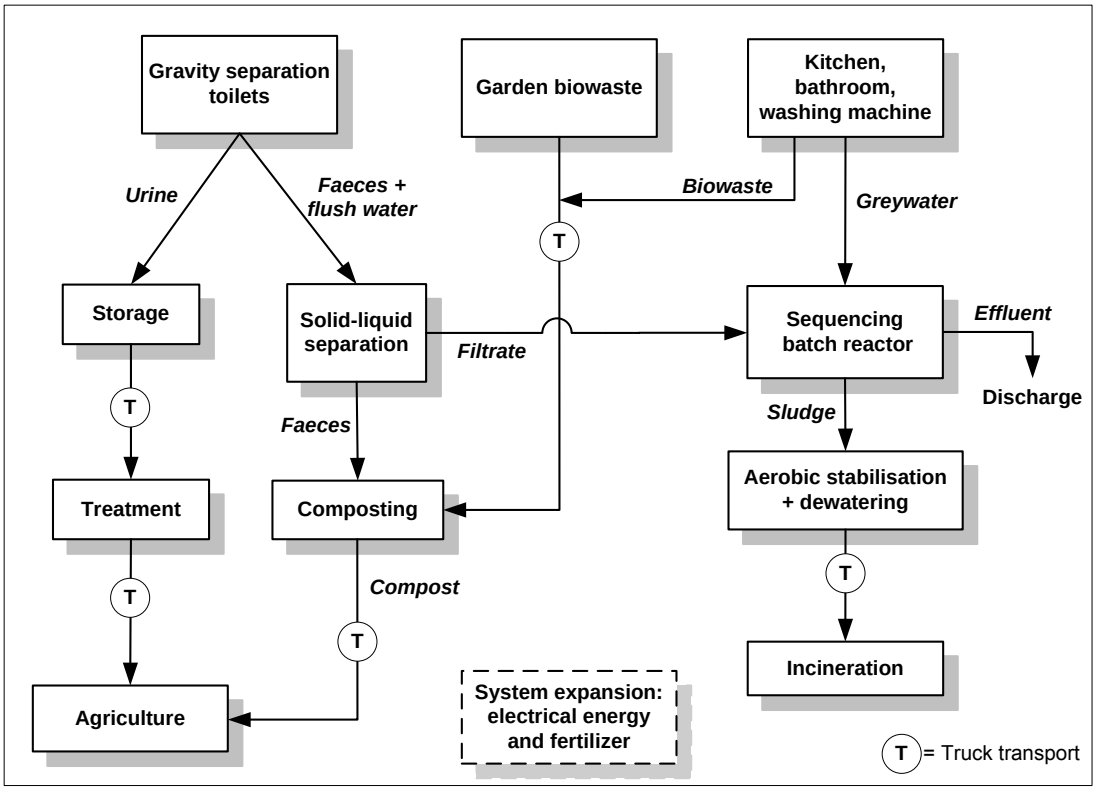

Figure 15: System setup of composting scenario with SBR (SC1)

Scenario SC3

The third composting scenario is targeting the reuse of treated greywater for toilet flushing (Figure 17). Therefore, the mixture of greywater and faeces filtrate is treated in a membrane bioreactor, which is a combination of an activated sludge process and a membrane stage for separation of sludge from the effluent. Due to the excellent effluent quality of the membrane separation step, the effluent can be reused for toilet flushing without further treatment. It is pumped back to the households via a separate pipe network and used for toilet flushing.

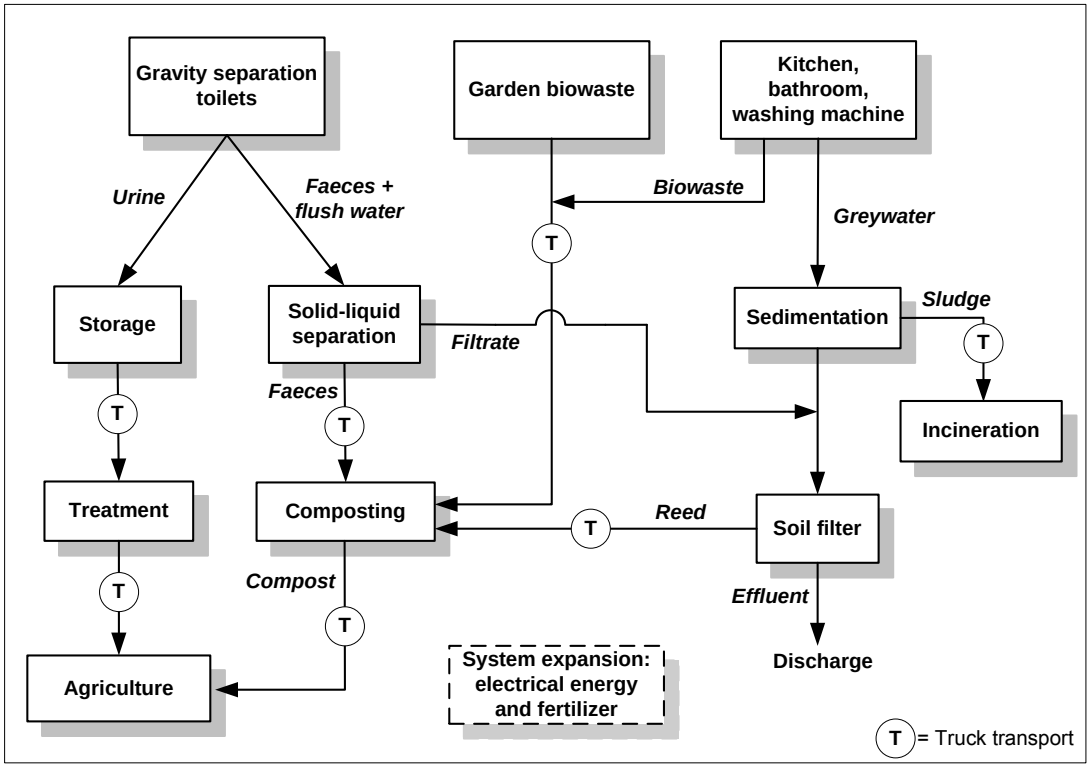

Figure 16: System setup of composting scenario with soil filter (SC2)

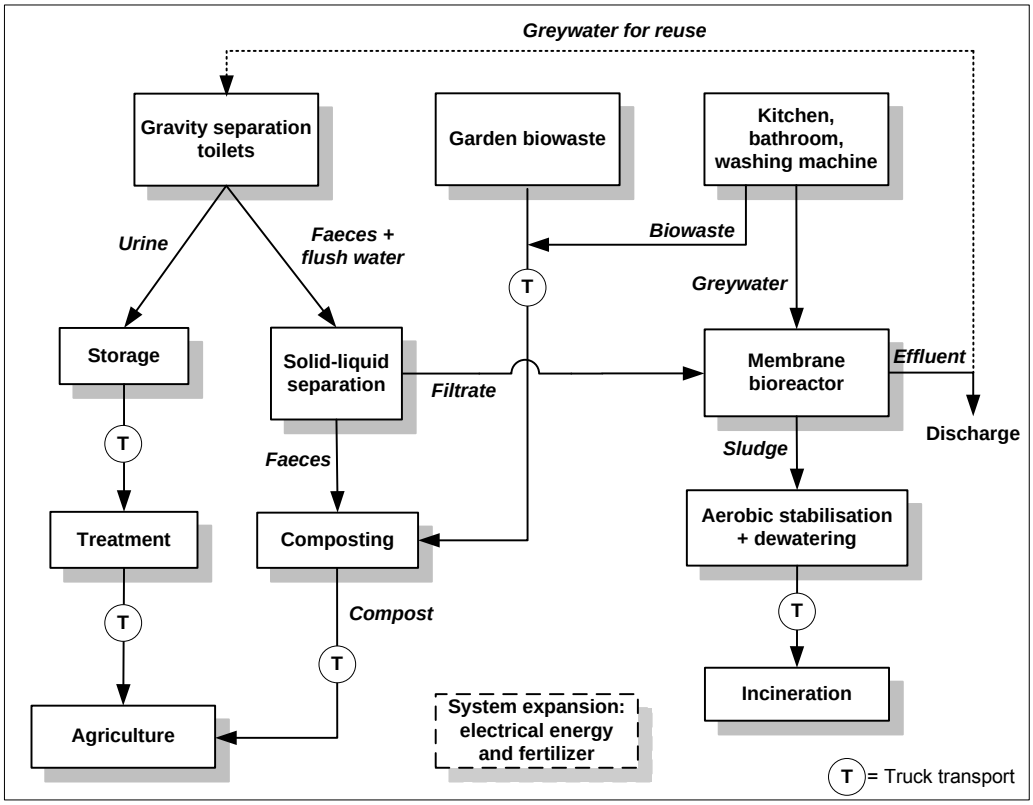

Figure 17: System setup of composting scenario with MBR and reuse (SC3)

3.6 System boundaries

The careful definition of system boundaries is an essential step in LCA methodology (ISO 14040, 2006). The choice of system boundaries determines the scope of the study and consequently the validity of its results. Moreover, the scope of a study has a major impact on the time and effort required for the Life Cycle Inventory. The availability of appropriate LCI data can be a crucial limit while defining the system boundaries of an LCA.

In order to make a specific statement about the investigated processes (i.e. the different sanitation systems) and to keep the LCA study manageable, system boundaries must be defined that run between (Guinée et al., 1993):

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

In the following chapter, important definitions with regard to the system boundaries of this study are specified and explained in detail.

3.6.1 System boundaries between the analysed economic system and the environment

The system boundary between the economic system under investigation and the environment is usually defined intuitively: Resources from the environment for consumption within the economic system cross this boundary ("input"), while emissions from economic processes to water, air, and soil pass the system boundary into the environment ("output").

However, processes in agriculture or landfill sites have characteristics that apply to ecosystems as well as to economic systems (Guinée et al., 2002). Agricultural soil is primarily used for production processes and thus can be seen as a part of the economic system. Consequently, emissions to agricultural soil stay within the economy and are not emitted into the environment. On the other hand, the soil itself is naturally a part of the environment and has a multitude of functions ("multifunctionality") including ecological functions, even if it is mainly used for agricultural purposes. Previous LCA studies of agricultural processes regard agricultural soil as part of the economic system right down to the groundwater table (Audsley et al., 1997) or as part of the environment to include long-term impacts on soil quality (Wegener Sleeswijk et al., 1996).

In this study, agricultural soil is regarded as a part of the environmental system. However, only heavy metals in fertilizers applied to agricultural soil are treated as emissions into the environment. The incorporation of nutrients and other substances into the crops is not further specified, because there is no relevance concerning the interests of this LCA study. The percentage of nutrient availability to the crop is considered with regard to the amount of mineral fertilisers that can be substituted by urine or other secondary fertilisers. Furthermore, primary emissions during fertilizer application (volatilization of NH_3, N_2O) are included in the inventory as well. Secondary emissions by subsequent processes in the soil (e.g. soil erosion, nitrification and denitrification, or migration of nitrate into the groundwater) are not considered in the inventory. These effects are difficult to quantify in a consistent way for the different types of fertilizers and are therefore excluded in this study, argueing that secondary emissions are comparable between all scenarios.

The landfill deposition of slags and ashes originating from incineration of biowaste or sewage sludge is not included in the scope of 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 slags are estimated to be negligible for the result of this LCA. A major fraction of these incineration residues is reused today as filling material in road construction (Hirschmann, 1999).

3.6.2 Multi function processes and recycling

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

- Combined production of several products at the same time (co-products)
- Combined waste air, wastewater and solid waste treatment as well as combined services such as transports in 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. LCA methodology recommends to avoid allocation, because it is often regarded as subjective (ISO 14044, 2006). Several procedures are available to avoid an allocation problem, for example dividing the respective unit processes in suitable sub processes or including the additional functions within the systems. The latter procedure is known as system expansion and is the preferred approach in many LCA studies (Curran, 2007). However, system expansion may lead to a larger and more complicated model that requires more LCI data.

System expansion for secondary functions

In this study, the primary function of all investigated systems is the collection and disposal of wastewater and biowaste. However, some scenarios deliver secondary functions such as the supply of organic fertilizers and energy. This allocation problem is overcome with the approach of system expansion by broadening the system boundaries (see chapter 3.4).

Allocation of co-products

Co-production applies mainly to energy and fuel production as well as to the production of caustic soda and chlorine during the supply of auxiliary materials. For these purposes, pre-allocated datasets from literature and Umberto® are used, which are widely accepted (e.g. IFU and IFEU, 2004; Boustead, 1998; Boustead, 1999a). In general, the allocation in these datasets is calculated on the basis of physical relationships (mass, energy, or molar proportions).

The combined treatment of solid waste in waste incineration plants and the treatment of wastewater in a municipal wastewater treatment plant are modelled widely on the basis of physical and chemical relationships. Emissions, residuals, and the subsidiary material demand of the waste incineration plant are calculated based on the waste composition and boundary conditions of operation (IFU and IFEU, 2004). For wastewater treatment, a substance flow model is used that has been specifically developed for LCA purposes at TU Berlin.

Recycling of construction materials

The recycling or disposal of construction materials after their prospected life cycle is defined depending on the type of material (Table 8). No recycling or system expansion is considered for concrete and vitrified clay, although these materials can potentially be recycled. Recycling of concrete is estimated to require approximately the same amount of energy that can be substituted with the recycled product (Baitz et al., 2004). Finally, concrete is often used as substitute for primary gravel, split, or sand in other applications.

Scrap metals made of steel or cast iron are supposed to be partially recycled, molten and converted to new metal products. This recycling procedure is taken into account by quasi-closed loop recycling, adopting actual recycling shares from Germany (BDSV, 2005). Plastic components are assumed to be incinerated for disposal, including the recovery of feedstock energy. Possible expenditures during recycling (transport, energy etc) are neglected here.

Table 8: 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, recycling share: 42% (BDSV, 2005)
Cast iron	Quasi closed-loop recycling, recycling share: 88% (BDSV, 2005)
Plastics	Incineration + recovery of feedstock energy

3.6.3 Considered life cycle phases and sub-systems

For the definition of considered life cycle phases and sub-systems, the Life Cycle Inventory is differentiated between processes of system operation and those associated with the production of capital equipment (infrastructure).

Operation

All relevant processes required for the operation of the analysed sanitation systems are included in the Life Cycle Inventory. Processes which are identical in all systems can be excluded without compromising the environmental evaluation. In general, each single process is linked again with all preceding und succeeding processes which are required for the production of energy, raw and auxiliary materials, or for the treatment and disposal of residuals. In particular the following life cycle phases are included:

- Domestic: Toilets (urination and defecation including toilet flushing), water consumption and pollution by personal hygiene and laundry washing; production of kitchen and garden biowaste
- Drinking water supply (energy demand for pumping and delivery)
- Conventional wastewater treatment (activated sludge process, including stabilisation of sewage sludge, dewatering and incineration or application in agriculture)
- Biowaste treatment (composting or co-incineration with municipal waste)
- Greywater treatment
- Faeces treatment: Dewatering and composting or vacuum drainage and anaerobic digestion
- Urine separation, storage, and treatment by ozonation
- Application of secondary fertilizers from urine, faeces, and biowaste
- Production and application of mineral fertilizers
- Production and supply of electric and thermal energy
- Transports

The inventory does not include inspection and maintenance, human labour or transport of workers. Although expenditures for maintenance may be significant, no suitable data is available for quantifying them and determine their environmentally relevant inputs and outputs.

Finally, drainage and treatment of stormwater, industrial or agricultural wastewater, and additional groundwater that is infiltrating through pipe leakages is not included in this study.

Capital equipment

The expenditures for the production of capital equipment (= infrastructure) are often small in relation to those for the operation of a technical process. Due to the relatively long lifetime of technical infrastructure, the time-related environmental impacts are negligible compared to operational flows and emissions. Consequently, the production of capital equipment is excluded in many LCA studies to minimize the complexity of the LCI model and the effort for data acquisition (Frischknecht et al., 2007).

However, the role of infrastructure in LCAs of wastewater treatment can be significant and is controversially discussed in literature. A case study of alternative sanitation systems in Sweden found that infrastructure had only a small and non-decisive impact on the environmental comparison (Tillman et al., 1998). Consequently, infrastructure is neglected in another Swedish LCA case study (Bengtsson et al., 1997). Other studies reveal a considerable influence of infrastructure in LCAs of wastewater treatment systems (Zimmermann et al., 1996). A recent review states that the share of impacts caused by capital goods manufacture is substantial in wastewater treatment, claiming the inclusion of infrastructure in these types of LCA (Frischknecht et al., 2007).

The present study includes both the necessary infrastructure and the operation of the respective sanitation systems in the assessment. The separation systems investigated in this study are characterized by the use of multiple pipe networks for the different wastewater flows, presumably leading to higher expenses for the production of pipes compared to the conventional system. It is therefore decided to include all capital equipment required for the construction of the different sanitation systems as far as the respective components are not identical in each system. Capital equipment of the background system (i.e. energy supply, any industrial production plant, road construction, etc.) is excluded from the inventory.

For the inventory of the infrastructure, it is assumed that the sanitation systems are integrated in a new urban development area together with the provision of other infrastructure (roads, power lines etc). The systems are also integrated inside the buildings during the initial construction (no rehabilitation of existing systems). The

following components are considered in the infrastructure inventory (for details cf. chapter 4.2):

- Materials for pipes for in-house and external drainage of combined wastewater, greywater, urine, and faeces (including inspection manholes)
- Excavation of trenches
- Conventional wastewater treatment plant
- Greywater treatment plants (SBR, soil filter, MBR)
- Storage tanks and pipe network for greywater reuse
- Urine collection and storage (pumping wells, pumps, storage tanks)
- Vacuum system
- Digester with stabilisation area for residual sludge and CHP plant
- Solid-liquid separators for faeces dewatering
- Service buildings for facilities of separation systems

The following infrastructure components of sanitation systems are excluded:

- Production of toilets (it is assumed that the associated expenses are roughly comparable for conventional and separation systems)
- Construction of composting plant: Both conventional and separation systems can include the composting of biowaste. Typically, the scale of a composting plant depends on the volume to be treated rather than on the mass (Bidlingmaier, 2000). As the expected volume variation of kitchen and garden biowaste is larger than the additional volume of faeces for composting, the scale of the composting plant is assumed to be the same in all systems. It is therefore excluded from the inventory.
- Facilities for urine treatment (ozonation plant)
- Electric measurement and control systems
- Drainage and treatment of stormwater and infiltrating groundwater: This is of particular relevance with regard to the size of the sewer and the wastewater treatment plant.
- Energy demand for the installation
- Human work and transport of workers

Figure 18 illustrates the procedure with regard to the consideration of capital equipment for sanitation systems and the background system which is not included in the inventory.

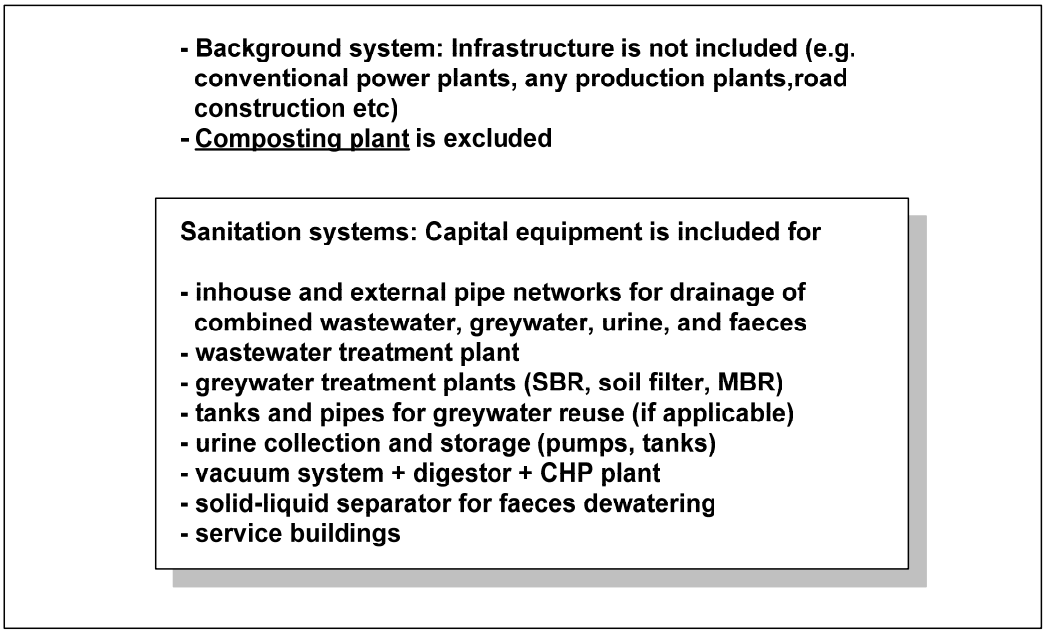

Figure 18: Consideration of capital equipment for sanitation systems and the background system

Cut-off criteria

In order to keep the complexity of the Life Cycle Inventory and the related substance flow model within reasonable limits, those processes and subsystems which are assumed to be not relevant for the environmental assessment are usually excluded from the inventory. Usually, a general cut-off criterion is defined in an LCA for this purpose. For example, material or energy flows falling below a certain limit in proportion to the total mass or energy flow (e.g. 1%) are cut off from further consideration.

In this study, no general cut-off criterion is defined due to the heterogeneity of the data quality in the LCI. 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 or amount of energy.

3.6.4 Considered elementary flows

According to ISO 14044, environmental interventions linked to unit processes are denoted as elementary flows. Elementary flows can be in the form of resources (e.g. crude oil), emissions to environmental compartments, or land use. In view of the large number of chemicals which are industrially produced or generated during combustion processes, the number of elementary flows potentially released to the environment can be extremely high. The latest datasets from LCI databases provide a large number of single substances in their inventory (e.g. > 1000 in Ecoinvent, 2007).

In this study, only selected elementary flows are considered with regard to the following criteria:

- elementary flows which in particular have a known significance in wastewater and biowaste management as well as in agricultural processes
- elementary flows of general environmental interest
- the selected elementary flows should quantify the considered impact categories in a sufficient way
- the respective elementary flows should be known for each process of the system (at least for the most important processes) in order to provide a symmetrical database
- elementary flows which are considered as possibly relevant but where no appropriate data is available have to be neglected (e.g. organic micro-pollutants)

Relevant elementary flows are defined before setting up the Life Cycle Inventory (Table 9). The listed elementary flows are assumed to be the most relevant for the ecological comparison. For the reference input flows of wastewater and biowaste, the inventory includes the distribution of organic material, nutrients, salts, and heavy metals to water, air, and soil. It should be again pointed out that organic micro-pollutants are not included in this assessment due to missing data for their occurrence in household wastewater and for their environmental relevance.

Datasets for material production, incineration plant, and background processes (transport, energy, mineral fertilizer) contain more emission flows of organic and inorganic substances to air (e.g. from combustion processes) and also a demand for mineral resources.

Table 9: Relevant elementary flows

Emissions to air	Emissions to water	Emissions to soil	Resources in geological deposits
CO2, CO, CH4	C, N, K, P, S	C, N, K, P, S	Lignite, Hard coal
N2O, NH3, NOx	in respective species	in respective species	Uranium
SOx			Natural gas
HCl, HF	As, Cd, Cr, Cu,	As, Cd, Cr, Cu,	Crude oil
PM 10, particles	Hg, Ni, Pb, Zn	Hg, Ni, Pb, Zn	Phosphorus
NMVOC		Fe, U	Lead
Benzene, BaP, PAH	Ca, Mg, Na, Cl	Ca, Mg, Na, Cl	Iron ore
Formaldehyde	AOX	AOX	Copper ore
PCDD / PCDF (TE)			Zinc
As, Be, Pb, Cd,			Bauxite, Sulphur
Cr, Co, Cu, Ni,			Nickel ore
Hg, Se, Th, Zn			Potash, Gypsum

Due to the simultaneous use of readily available datasets from LCI databases for background processes and own process models for the sanitation systems, the symmetry of the LCI data is not completely consistent. Datasets for background processes generate up to 130 different elementary flows of emissions (e.g. during the production of electric energy), but most of them are not decisive for the impact assessment and are neglected in the further documentation of this study. Material flows in the system (wastewater, biowaste etc) and related process emissions from wastewater treatment are decomposed into their elementary composition as far as suitable data is available (Table 10).

Table 10: Material flows and their elementary composition

Material flows in the system	Elementary composition
Reference input flows (Combined wastewater or urine, faeces, greywater, biowaste)	C, N, K, P, S Ca, Mg, Na, Cl Cd, Cr, Cu, Hg, Ni, Pb, Zn AOX
Mineral fertilizers	N, K, P, S, Ca,Cl As, Cd, Cr, Cu, Hg, Ni, Pb, U, Zn
Precipitation chemicals	Fe, S, Cl
Construction material	No elementary decomposition is considered
Electric, thermal and mechanical energy	

3.6.5 Geographical and temporal scope

The present study is a prospective LCA, because the separation systems under investigation have not been realized in large scale. However, the process design and technology of the different scenarios is available at the time of this study or will be available in the near future. For definition purposes, it is assumed that all sanitation systems are operated in Germany in 2007.

Geographical and temporal scope: Germany 2007

The process technology reflects available state-of-the-art technology of the year 2007. Still, several assumptions have been adopted for process performance and emission data, because respective data has not been generated or published.

The background data of this study (energy, transport, materials) mainly refers to conditions in Germany during 1990 – 2000 as available in the UMBERTO® database (IFU and IFEU, 2005). Updates of datasets are identified accordingly in the inventory. 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.

Impact assessment

Environmental interventions are aggregated without accounting for time and exact location of emissions. With regard to impact assessment, spatial differentiation on a regional level (Germany) is adopted if possible. The indicators used in the impact assessment mostly assume a defined time horizon for the impacts of emissions (Guinée et al., 2002). For global warming potential, a time horizon of 100 years is considered.

Validity of the results

This study reflects conditions in Germany concerning available process technology and performance, spatial differentiation in impact assessment, and interpretation of LCIA results (normalisation and weighting). The outcomes of this study may also be valid for other geographical regions with similar technological development (e.g. industrialized countries of Western Europe). However, regional conditions such as the local power mix or legal requirements for effluent discharge (= discharge standards, disinfection, micro-pollutants, recycling quotas etc.) can have a significant impact on the results of the assessment.

3.7 Data quality of Life Cycle Inventory

General remarks

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 combined sewage without stormwater in a settlement with 5,000 inhabitants in Germany in the year 2007). However, this optimum data quality cannot be reached in most cases due to lack of adequate time and financial means for data collection. Sometimes, adequate data with high quality may be existent, but it is confidential or simply not published by the operator. Hence, there is always a trade-off between the data quality of the LCI 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. recalculation or upscaling) to fit into the specific boundary conditions of the LCA study. Where no appropriate data is available, qualified assumptions have to be made to allow for a coherent inventory analysis. Thus, the assessment of data quality in an LCA study is an important indicator for its representativeness and hence for the validity and significance of its conclusions.

Data quality of this study

The present study is designed to assess the environmental impacts of different sanitation systems with LCA methodology. Modelling the construction and operation of these systems for the Life Cycle Inventory requires an extensive collection and review of data. To keep the study effort within reasonable limits, there has to be a compromise between the level of detail and data quality on the one hand and the effort and time required for data collection on the other hand.

Moreover, separation systems have not yet been implemented in a larger settlement in Germany. Thus, high quality primary data of full-scale plants is not available for most of the processes of separation scenarios. Secondary data from pilot-scale tests or laboratory experiments has to be used to generate a defensible basis for the LCI. Upscaling this data for a relatively large settlement is a delicate task and requires reasonable assumptions. The prospective nature of this study – assessing systems that are not in operation yet – can justify the use of data which may not be representative for large-scale systems. However, the origin and the quality of LCI data have to be clearly identified to show the limitations of this study. The following chapter gives a short overview about data quality of the LCI, necessary assumptions and methodological limitations of this study.

3.7.1 Data quality of the present LCI

LCI data of core processes (Table 11)

The composition of the different wastewater flows is determined from an extensive review of respective literature. Although the specific composition can vary depending on local user pattern and habits, the chosen values are thought to represent a reasonable average of the German population. Process data for the conventional sanitation system is based on an elaborate LCA model of an activated sludge plant. The process model is originally designed for larger plants (> 10000 inhabitant equivalents), but is adjusted to the present dimension in terms of elimination ratios and energy demand.

The processes of separation systems are mainly based on experiences from pilot projects, pilot plants, and laboratory experiments. Pilot projects include Berlin-Stahnsdorf (~ 35 inhabitants), Lübeck-Flintenbreite (108), Freiburg-Vauban (30), and several Swedish sites (160+160). Results from these pilot studies have to be adapted to

the conditions of this study. However, data for urine separation, storage and application as well as for greywater treatment in soil filters and membrane bioreactor is regarded as appropriate to reflect conditions in a larger settlement. Ozonation of urine and co-digestion of faeces and biowaste have only been investigated in small-scale laboratory experiments, so that LCA data for these processes may be different in full-scale applications. Data for the digestion process is further adopted from an LCA study of biowaste treatment, assuming comparable energy demand and transformations of elemental composition.

The processes of faeces composting with biowaste and the aerobic stabilisation of digester residual from faeces co-digestion have not been investigated in terms of emission data, so that LCI data from biowaste treatment is used here. The process of faeces dewatering prior to composting has not been technically realized in a satisfactory way, so that the efficiency of the process and the quality of the faeces filtrate have to be estimated.

LCI data of background processes (Table 12)

Data for the supply of energy and transport expenditures is adopted from approved databases. The electrical energy data is updated with national data of the energy mix in Germany in 2003. Transport distances in the system are heavily depending on local conditions, so that reasonable estimates are used and their influence is quantified in sensitivity analysis. The construction inventory is set up with a high level of detail in cooperation with experts for the design of sanitation systems. LCI data for the supply of construction materials is calculated with a combination of literature data and datasets from Umberto®. The LCI datasets for the production of mineral fertilizers are relatively outdated, but more recent comprehensive datasets have been unaccessible for this study. Heavy metal contents of mineral fertilizers is based on information of 1992 and should be updated (the effect of more recent, but not representative data is calculated in sensitivity analysis). Emissions during fertilizer application are estimated according to emission inventories for the European agriculture.

3.7.2 Important assumptions and limitations of this study

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.
- Sum parameters (COD, NMVOC, etc.) allow only a restricted evaluation of the ecological significance of emissions

Specific limitations

- Potentially harmful organic micropollutants (pharmaceuticals, etc.) in human excreta are not evaluated due to missing data of occurrence and toxicity
- The beneficial effects of organic carbon in compost (e.g. for soil fertility) are not quantified here and are neglected regarding fertilizer substitution. This constraint may lead to an underestimation of the fertilizer substitution potential of separation systems.
- The migration of nitrate into the groundwater and associated impacts are not considered. It is assumed that mineral and secondary fertilizers exhibit comparable leaching rates for nitrate
- The constructional design of wastewater systems, the operational expenses and the associated environmental impacts depend heavily 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.).
- Only a limited set of environmental interventions and impact categories are included (e.g. no odour exposure) because of lack of data and missing symmetry of data
- The results are strongly influenced by the system expansion with the production of mineral fertilizers. However, the LCI datasets for fertilizer production and heavy metal content are relatively old

Table 11: Origin and quality of inventory data for core processes

Processes	Main data sources	Data origin	Reference period	Data quality	Remarks
Composition of wastewater flows	Average values from literature	Literature	1995-2000	High	Literature reviews and case studies
General operational parameters*	Peter-Fröhlich et al., 2007 Otterwasser, 2005	Pilot projects	2003-2006	High	~ 35 inhabitants ~ 108 inhabitants
Activated sludge plant/SBR	TU Berlin	Literature + national surveys	2003	High	LCA model by TU Berlin
Membrane bioreactor	Pinnekamp and Friedrich, 2006 Peter-Fröhlich et al., 2007	Review Pilot plant	2000-2006 2006	Moderate High	MBR characteristics Treatment of greywater
Soil filter	Peter-Fröhlich et al., 2007 Otterwasser, 2005	Pilot plants	2003-2006	High	~ 35 inhabitants ~ 108 inhabitants
Incineration	IFU and IFEU, 2004 + literature	Operational data of full-scale plants	1990-2000	Moderate	Waste incineration plant modified for sludge incin.
Composting	Vogt et al., 2002	LCA study	2000	Low	LCA study of biowaste composting
Faeces co-digestion	Vogt et al., 2002 Otterwasser, 2005 + literature	LCA study Lab experiments	2000 2003-2007	Low Moderate	Biowaste digestion Blackwater digestion
Urine separation	Stockholm Vatten, 2000 Peter-Fröhlich et al., 2007	Pilot projects	1995-2000 2006	High Moderate	~ 320 inhabitants ~ 35 inhabitants
Urine treatment	Stockholm Vatten, 2000 Escher et al., 2006	Pilot experiments Lab experiments	1995-2000 2006	High Moderate	Storage Ozonation
Urine application	Stockholm Vatten, 2000 Peter-Fröhlich et al., 2007	Field tests Field tests	1995-2000 2006	High High	Availability of urine-N + N emissions

* flush water demand for toilets, energy demand for vacuum system, faeces dewatering etc

Table 12: Origin and quality of inventory data for background processes

Datasets	Main data sources	Data origin	Reference period	Data quality	Remarks
Energy electrical	Fritsche et al., 2001 Frischknecht et al., 1996 BMWi, 2005	Database LCI National data	1990 + 1998 1995 2003	High Moderate Moderate	Power plant technology Some emission data Energy mix
Energy thermal	Fritsche et al., 2001	Database	1990	Low	Power plant technology
Transport by truck	TREMOD (Knörr et al., 1997)	Database	1996	High	Emission model
Transport distances	Assumptions based on literature	LCA studies	1997-2000	Moderate	Dependent on local conditions
Materials					
plastics	Boustead, 1999a	LCI	1995	Low	APME data (old)
concrete	Frischknecht et al., 1996 + literature	LCI	1995 + 2002	Moderate	Updated LCI
limestone	Patyk and Reinhardt, 1997	LCI	1995	Moderate	
metals	Frischknecht et al., 1996 + literature	LCI	1990-2000	Low	Updated LCI (old)
flocculant	Manufacturer information + literature	Literature	2000	Moderate	
Construction inventory	Peter-Fröhlich et al., 2007	Consultants	2006	High	Exemplary system layout
Industrial fertilizer production	Patyk and Reinhardt, 1997 Gaillard et al., 1997 Boysen, 1992 Drescher-Hartung et al., 2001	LCI LCI Literature Market data	1990-1995 1992 1992 1998/99	Moderate Moderate Low Moderate	Resources + emissions Water emissions Heavy metal contents Market shares
Fertilizer application	EMEP/CORINAIR, 2004 Borken et al., 1999	Review LCI	1992 1995	Moderate Moderate	Emission inventory Tractor emissions

3.8 Life Cycle Impact Assessment

The procedure of Life Cycle Impact Assessment (LCIA) aims at connecting the results of the Life Cycle Inventory (e.g. extraction of resources, emissions) to their potential environmental impacts on the basis of impact pathways. Impact pathways consist of linked environmental processes, and they express the causal chain of subsequent effects originating from an emission or extraction. It has to be kept in mind that LCIA cannot determine whether actual effects occur, but it delivers information about potential environmental impacts or hazards.

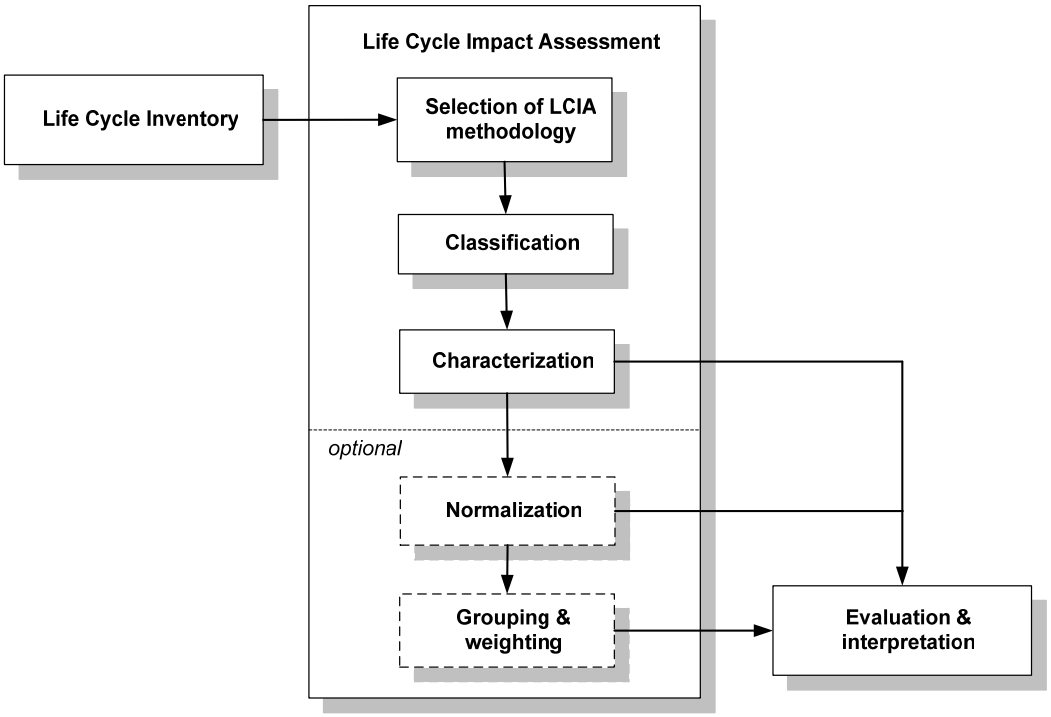

Figure 19: Elements of Life Cycle Impact Assessment (ISO 14040, 2006)

In general, LCIA consists of the following steps (ISO 14044, 2006) (Figure 19):

1. **Selection of LCIA methodology:** Suitable impact categories, related indicators and characterization models have to be chosen in relation to the goal and scope of the study. This can be an iterative process.

2. **Classification:** LCI results are assigned to the impact categories for which they are relevant.

3. **Characterization:** Extractions and emissions are characterized by scientifically derived factors. Aggregation of all relevant LCI flows multiplied with their characterization factors results in the overall category indicators.

4. **Normalization (optional):** In order to gain a better understanding of the relative size and importance of a category indicator result, a normalization step can optionally be integrated. Each indicator is calculated in relation to reference information (e.g. total impact in a reference area on a per capita basis) as a benchmark.

5. **Grouping and weighting (optional):** Indicators can be grouped according to the goal and scope of the study, based either on a nominal basis (sorting) or on value-choices (ranking). Weighting includes the conversion of indicator results of different impact categories by using numerical factors based on value-choices. These weighted indicators can be evaluated individually or aggregated across impact categories.

Finally, the LCIA results are evaluated and interpreted with regard to the goal and scope of the study. This includes the identification of significant issues or system parameters based on the LCI and LCIA results, and an evaluation of completeness, sensitivity, and consistency of the study. With a sensitivity analysis for the LCI and LCIA phase, the reliability of the final results can be assessed by determining how they are affected by uncertainties in LCI data, LCIA methodology, or other relevant parts of the study. Ultimately, conclusions from the LCA study can be drawn while identifying possible limitations of the study and making recommendations for the audience.

3.8.1 Selection of LCIA methodology

A number of different methodologies for Life Cycle Impact Assessment have been developed in recent years according to the requirements of ISO 14040/44. These LCIA methods include among others the CML guide (Guinée et al., 2002), Ecoindicator 99 (Goedkoop and Spriensma, 2000), EDIP (Hauschild and Potting, 2003), IMPACT 2002+ (Jolliet et al., 2003a), TRACI (Bare et al., 2003) and LUCAS (Toffoletto et al., 2007). In general, Life Cycle Impact Assessment methods can be grouped into two families (Jolliet et al., 2004):

A) Classical impact assessment methods that stop quantitative modelling before the end of the impact pathways and link LCI results to so-defined midpoint categories (“midpoint-oriented”)
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, i.e. the damage to human health, ecosystem quality etc. (“endpoint- or damage-oriented”)

The difference between these types of LCIA lies in the modelling depth of the cause-effect chain (Figure 20). Both methods have specific benefits and drawbacks, and the choice for a certain LCIA method depends on the specific goal and scope of the LCA study. Recently, a new LCIA method providing midpoint and endpoint characterization factors has been published (Goedkoop et al., 2009), combining methodologies of CML and Ecoindicator. Thus, both modelling depths are available in the same methodological framework.

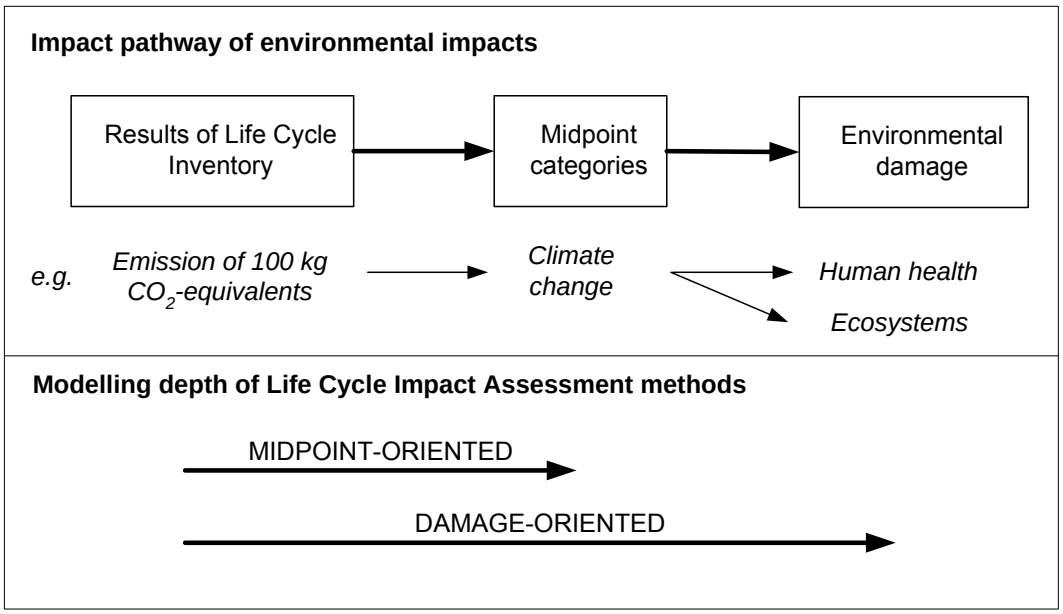

Figure 20: General structure of methods for Life Cycle Impact Assessment

For the impact assessment in this study, a well-established *midpoint-oriented approach* is used based on the methodological LCA guide developed at the Centrum voor Milieukunde at Leiden University (CML method, Guinée et al., 2002). The environmental impacts of the respective sanitation systems can be compared regarding each impact category for itself. The results are more transparent and comprehensible. A damage-oriented approach would produce results which are easier to compare as there are usually only a small number of endpoint categories, but the second step of impact modelling (from midpoint to endpoint categories) would lead to higher uncertainties of the results. Furthermore, the evaluation of differences in endpoint categories between the compared sanitation systems is more difficult, as the cause of the difference is not necessarily traceable.

The focus of the present study is not to give a conclusive answer which sanitation system has the lowest total environmental impacts, but to identify the benefits and drawbacks of each system by a systematic analysis of the specific impacts. Therefore, a midpoint-oriented LCIA approach is more appropriate for the specific goals of this study.

Impact categories and category indicators

The CML method distinguishes three sets of impact categories, depending on the environmental relevance in relation to LCA and the availability of adequate characterization methods (Guinée et al., 2002):

1) Group A comprises the baseline impact categories which are typically included in many LCA studies
2) Group B comprises study-specific impact categories which may be included depending on the goal and scope of the study and whether appropriate data are available
3) Group C comprises several categories for which no baseline characterization method is available and which require further research and elaboration

It is decided that only baseline impact categories (Group A) are considered in this LCA study. These categories represent the most important environmental impacts of sanitation systems and are mainly based on a well-established methodology. The specific categories of Group B and C are either not applicable in the context of sanitation systems (e.g. impacts of ionising radiation, noise, waste heat, desiccation etc) or the required LCI data is inconsistent or not available (loss of life support function and biodiversity due to land use, odour etc).

The CML method includes 11 baseline impact categories, of which 7 are applied in this LCA study (Table 13). Four categories have been excluded, basically because the relevant inventory data for these categories is not available or largely inconsistent in the LCI (land use, stratospheric ozone depletion, photo-oxidant formation). For ecotoxicity, it is decided to reduce the impact assessment to the effects on terrestrial and freshwater aquatic ecosystems (excluding marine ecosystems and sediments).

The baseline impact categories of the CML method are complemented by a specific impact category related to the demand of non-renewable energy resources (VDI, 1997). The rising demand for finite energy resources is a major concern for society today, and precise information on this sensitive issue should be included in each assessment of sustainability.

Table 13: Categories for Life Cycle Impact Assessment

Impact category	Description of the environmental impact
Resource-related	
Demand of non-renewable energy sources	consumption of finite fuels for energy production, i.e. fossil and nuclear fuels
Depletion of abiotic resources	consumption of finite non-living natural resources, including fossil fuels and mineral resources (ores etc)
Emission-related	
Climate change	impact of human emissions on the radiative forcing (i.e. heat radiation absorption) of the atmosphere
Acidification	release of protons in surface waters and soil due to oxidation and hydrolysis of atmospheric gases
Eutrophication	excessive levels of macronutrients (nitrogen and phosphorus) and degradable organic carbon in aquatic and terrestrial ecosystems, causing oxygen deficiency through elevated biomass production and decomposition
Human toxicity	impacts of toxic substances present in the environment on human health.
Freshwater aquatic ecotoxicity	impacts of toxic substances on freshwater aquatic ecosystems
Terrestrial ecotoxicity	impacts of toxic substances on terrestrial ecosystems

3.8.2 Classification

In a next step, the emissions and extractions from the Life Cycle Inventory are attributed to the impact categories to which they are contributing (Table 14). The LCI data in this study is heterogeneous in quality and in the number of considered material flows. Whereas LCI datasets for the background system can comprise more than 100 different input and output flows (e.g. supply of electric energy, steel production etc), the inventory for the core processes is limited to the elementary flows. For reasons of clarity, the classification of emissions and extractions is limited to the substance flows which are expected to be relevant for the impact assessment. The contribution of substance flows which are not listed in Table 14 to the respective impact categories is estimated to be < 0.1% of the total impact category.

Table 14: Classification of emissions and extractions to their related impact categories

Impact category	Relevant emissions/resources
	(R: resources, W: emission to water, A: emission to air, S: emission to soil)
Depletion of abiotic resources	(R): lignite, hard coal, natural gas, crude oil, uranium, raw phosphate, lead, iron/copper/nickel/chromium ore, zinc, bauxite, sulphur, potash
Demand of non-renewable energy sources	(R): all forms of primary energy from fossil and nuclear resources
Climate change	(A): CO_2, CH_4, N_2O
Acidification	(A): NH_3, NO_x, SO_2, HCl, HF
Eutrophication	(W): P species, N species, COD/TOC (A): NH_3, NO_x
Human toxicity	(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, CH_2O, BaP, PAH, PCDD/PCDF (S): Cd, Cr^{III+IV}, Cu, Hg, Ni, Pb, Zn
Aquatic Ecotoxicity Terrestrial Ecotoxicity	(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

Note: listed are only relevant emission flows (> 0.1% contribution to impact category)

3.8.3 Characterization

All relevant emissions or extractions from the Life Cycle Inventory are characterized by scientifically derived factors ("characterization factors") which describe the respective environmental impact in relation to a fixed reference, e.g. the emission of a certain reference substance with a known effect in this category. Thus, all environmental impacts in a certain impact category can be aggregated to one indicator score with a common unit.

For the impact assessment, a set of seven indicators from CML has been selected together with the cumulative energy demand of VDI (Table 15). In sensitivity analysis, four indicators from EDIP and IMPACT methodology for eutrophication and ecotoxicity are calculated and compared to the baseline indicators to reveal possible influences of the indicator choice on the results.

Table 15: Midpoint indicators for Life Cycle Impact Assessment

Indicator	Abbr (*Unit*)	Remarks	LCIA method
Resource-related indicators			
Cumulative energy demand	CED (*MJ*)	fossil + nuclear fuels	VDI
Abiotic depletion potential	ADP (*kg Sb-eq*)		CML
Emission-related indicators			
Global warming potential	GWP (*kg CO_2-eq*)	factors for GWP_{100a} (IPCC, 2001)	CML
Acidification potential	AP (*kg SO_2-eq*)	regional factors for $NH_3/NO_x/SO_2$ in air	CML
Eutrophication potential	EP (*kg PO_4-eq*)	regional factors for NH_3/NO_x in air	CML
Human toxicity potential	HTP (*kg DCB-eq*)	fate and exposure included	CML
Freshwater aquatic ecotoxicity potential	FAETP (*kg DCB-eq*)	fate and exposure included	CML
Terrestrial ecotoxicity potential	TETP (*kg DCB-eq*)	fate and exposure included	CML
Indicators for sensitivity analysis			
Terrestrial eutrophication	TEU (*m² UES*)	regional factors for NH_3/NO_x in air	EDIP
Aquatic eutrophication of inland waters	AEU (*kg NO_3-eq*)	regional factors for N and P in water	EDIP
Aquatic ecotoxicity	AET (*kg TEG*)	fate and exposure included	IMPACT
Terrestrial ecotoxicity	TET (*kg TEG*)	fate and exposure included	IMPACT

DCB: 1,4-dichlorobenzene, TEG: triethylene glycol, UES: unprotected ecosystem
Sources: Guinée et al., 2002 (CML); Hauschild and Potting, 2003 (EDIP); Jolliet et al., 2003a (IMPACT)

The calculation of characterization factors is based on distinct scientific models for the different indicators. The following part describes in short the underlying scientific models for the indicators applied in this study. The characterization factors which have been used for the impact assessment are listed for each indicator in annex 12.2.

Fate and exposure
In general, emissions into the environment can cause impacts on a local, regional, or global scale. While global impacts are usually not dependent on the point of emission, local and regional impacts can vary between different media, different regions, and different ecosystems. Local impacts are determined by the sensitivity of the local environment and local concentrations, which depend on the environmental fate of the substance and the quantity of emissions. Therefore, the impact assessment of emissions causing local and regional effects should take the fate of the substances into account by adding a factor for fate and exposure to the calculation of the category indicator:

$$S^m = \sum_i F_i^m \cdot E_i^m \cdot M_i^m$$

with *S = category indicator*
m = medium, i = substance
F = fate and exposure factor (≤ 1)
E = effect factor
*F*E = characterization factor*
M = mass emitted

However, the calculation of consistent factors for fate and exposure and effect is dependent on the availability of appropriate data from local or regional ecosystems and adequate scientific models for transport, fate and effect of the emissions. For the majority of indicators in this study, these factors are available and are thus applied if possible. Other indicators are still lacking consistent models, so that maximum characterization factors have to be used. Thus, these factors represent a “maximum” potential of environmental impact without consideration of local or regional conditions.

Resource-related indicators

Abiotic depletion potential (ADP)
The calculation of this indicator relates the demand for each specific resource to the reference substance of antimony (Sb). The characterization factors are based on the ratio of the ultimate reserve of each resource and its annual global de-accumulation, relative to those for antimony (Guinée et al., 2002). It has to be noted that exact data of reserve and consumption is subjected to a considerable level of uncertainty for many resources (exploration of new deposits, cost effectiveness of exploitation etc).

Cumulative energy demand (CED)
This indicator directly quantifies the amount of primary energy which correlates to the amount of fossil and nuclear fuels that are consumed for a particular process (VDI,

1997). It is a well-established indicator which has been determined for a large number of products and services, thus making it a simple and effective measure for the total energy demand. It has been shown that cumulative fossil energy demand can be a good screening indicator for the overall environmental performance of certain products or systems (Huijbregts et al., 2006).

Emission-related indicators

Global warming potential (GWP)
This indicator quantifies the potential of greenhouse gases to contribute to global warming. It is related to CO_2 as a reference substance. The characterisation factors are based on an elaborate model developed by the Intergovernmental Panel on Climate Change (IPCC, 2001) for a time horizon of 100 years. CO_2 emissions from renewable fuels (e.g. from biogas combustion) are not accounted for this indicator, neither are gases with indirect effects on climate change.

Acidification potential (AP)
Factors for acidification describe the potential of the emitted gases to form H^+ ions, related to the reference substance of SO_2. The sensitivity of the region in which the acidifying substances are emitted and the subsequent fate are considered for the three main acidifying air pollutants (NH_3, NO_x, SO_2) with a dispersion model for long-range transport of air pollutants (EMEP) and the acidification model RAINS-LCA (Huijbregts, 1999a). The effect factors are based on a ratio of deposition and critical load, taking into account only emissions above a specific threshold (baseline emissions: 1995) and Germany as the area of emission.

Eutrophication potential (EP)
This indicator describes the eutrophication of aquatic and terrestrial ecosystems due to aquatic emissions of nitrogen, phosphorus, and degradable organic carbon and air emissions of NH_3 and NO_x. Usually, phosphorus is the limiting macronutrient in freshwater ecosystems, while nitrogen is limiting in marine and terrestrial ecosystems. The effect factors for eutrophication are generally based on the Redfield ratio which represents the average chemical composition of aquatic phytoplankton (C : N : P = 106 : 16 : 1 on a molar basis). Hence, they should be seen as a highest potential contribution to biomass growth rather than an effective effect on biomass growth in the respective ecosystems. While the fate of aquatic emissions is not taken into account (fate factor = 1), site-dependent characterisation factors for air emissions of NH_3 and NO_x are calculated for Germany with the RAINS-LCA model (Huijbregts, 1999a). The reference substance for the characterization factors is PO_4.

Human toxicity potential (HTP)
Characterization factors for human toxicity have been calculated including degradation of substances and intermedia transport with the multi-media fate, exposure and effect model USES-LCA (Huijbregts, 1999b). This model includes five spatial scales, six environmental compartments, and six exposure routes. The factors are related to 1,4-dichlorobenzene as a reference substance, considering a global scale and an infinite time horizon ($HTP_{global, \infty}$). The selection of an infinite time horizon results in a high influence of persistent heavy metals on this indicator, while degradable organic chemicals are less dominant.

Terrestrial and freshwater aquatic ecotoxicity potential (TETP, FAETP)
The calculation of characterization factors for these indicators is basically similar to that described for the human toxicity potential. Degradation of substances and intermedia transport is considered using the multi-media fate, exposure and effect model USES-LCA (Huijbregts, 1999b). The characterization factors relate to 1,4-dichlorobenzene as a reference substance. The effect factors are calculated from the ratio of Predicted Environmental Concentration (PEC) and Predicted No Effect Concentration (PNEC) for aquatic and terrestrial ecosystems (= risk characterization factor).

Indicators for sensitivity analysis

Terrestrial eutrophication (TEU)
This indicator calculates the impact of nitrogen gases NH_3 and NO_x on the eutrophication of terrestrial ecosystems. Critical loads, fate and exposure are modelled based on the RAINS model for emissions in Germany and the predicted emission levels of the year 2010 (Hauschild and Potting, 2003). The impact is expressed as the area of terrestrial ecosystem that exceeds the critical load (m² of unprotected ecosystem or UES).

Aquatic eutrophication of inland waters (AEU)
This indicator quantifies the impact of nitrogen and phosphorus emissions on the eutrophication of aquatic ecosystems of inland waters (rivers and lakes). The influence of degradable organic carbon and all air emissions are excluded in this method. Fate and exposure of all relevant emissions to water is modelled in high spatial resolution with the CARMEN model (Hauschild and Potting, 2003). Characterization factors relate to NO_3 as a reference substance and Germany as the place of emission and impact. The method is designed to include the effect of nitrogen and phosphorus input on agricultural soil and subsequent leaching/run-off in aquatic ecosystems. However, the contribution of nutrient application in agriculture is excluded here for reasons of consistency to the other eutrophication indicators. Furthermore, the impact of

agricultural emissions would be comparable for all investigated scenarios as equal amounts of nutrients are applied in each scenario by definition.

Terrestrial and aquatic ecotoxicity (TET, AET)
This method for the assessment of ecotoxicity is based on a different approach. The fate factor is defined as the residence time of a substance in a "global" freshwater or terrestrial ecosystem divided by a dilution factor (Jolliet et al., 2003a). The overall residence time is determined by modelling volatilization, sedimentation, and degradation processes. Intermedia transfer is included by fractions transferred between air, soil, and water. The effect factor is calculated as the inverse PNEC value. The result can be interpreted as an area polluted up to the PNEC over a certain time determined by the fate of the substance ("critical surface time"). For communication purposes, the impacts are related to triethylene glycol as a reference substance. This method was originally developed for non-polar organics and has been adapted to include metals.

Review of toxicity assessment in LCIA

The characterization of toxic impacts on human health or ecosystems is a complex task. Toxic effects of emissions into the environment are dependent on the fate of the substance, threshold phenomena, the exposure time (acute vs. chronic) and pathways, and potential cumulative effects for mixtures of toxic substances. For ecotoxicity, the impact of toxic substances further depends on the affected species and characteristics of the ecosystem.

Hence, the modelling of toxic impacts for LCIA is still under development. Present state-of-the-art models take into account fate and exposure and have different approaches to calculate effect factors. However, the calculated characterization factors for human and ecotoxicity can still be affected with high uncertainty. For example, characterization factors of the CML method which have been calculated for different time horizons (20/100/500a or infinite) show differences of up to 6.5 orders of magnitude for some toxicity potentials of heavy metals (Huijbregts, 2000). This is particularly attributed to the persistence of heavy metals and the weak modelling of their speciation and intermedia transport without spatial differentiation of the environmental conditions.

The problem of a realistic assessment of toxic impacts in LCIA is intensively discussed in the international LCA community (e.g. Guinee et al., 2004). A major task is seen in calculating improved characterization factors for heavy metal emissions by extending available models to include effects of speciation, persistence, essentiality, and bioavailability along with further elaboration of fate and effect models (Ligthard et al., 2004). Until then, the deficiency of current LCIA methods in terms of toxicity assessment should be clearly communicated.

Due to the availability of different methods for toxicity assessment in LCIA, the choice of the evaluation method may have a significant influence on the outcome of the

LCIA. In comparative studies, different LCIA methods for aquatic ecotoxicity have been applied to a fixed Life Cycle Inventory, revealing a distinct influence on the contributions of specific substances to the overall indicator (Schulze et al., 2001; Gloria et al., 2006). Another recent work compares different approaches to calculate toxic effect factors, concluding that none of the existing approaches can be recommended as optimal in their present form (Larsen and Hauschild, 2007). A combined model for toxicity assessment of chemicals (USEtox) has been presented recently (Rosenbaum et al., 2008), but it is not yet ready for application.

In view of these difficulties with toxicity assessment in LCIA, the results of human and ecotoxicity in this study should be regarded with care. The exclusion of organic micropollutants from the elementary flows will result in a dominant influence of heavy metals on the toxicity indicators. Hence, the major uncertainties associated with the toxicity assessment for heavy metals are reflected in these indicators. One should keep that in mind while interpreting the toxicity indicators and their influence on the overall comparison. Apart from the baseline indicators from CML, toxicity indicators from IMPACT methodology are applied in sensitivity analysis to reveal possible influences of indicator choice on the results.

3.8.4 Normalization

Normalization is an optional step for LCA that can be applied for a better understanding of the relative magnitude for each indicator result (ISO 14044, 2006). It is commonly used for checking for inconsistencies, providing information on the relative significance of the indicator results, and as a first step in weighting. A normalized score for a certain impact category is obtained by determining the ratio of the category indicator result of the product system and that of a reference system:

$$I_{i,norm} = \frac{I_{i,system}}{I_{i,reference}}$$

with $I_{i,\,norm}$ = *normalized value of indicator i*
$I_{i,\,system}$ = *score of indicator i for the product system*
$I_{i,\,reference}$ = *score of indicator i for the reference system*

The reference systems applied for normalization purposes in LCA are typically defined as global or regional annual values, e.g. the world in a certain year or the population of a certain country in a certain year. The indicator values for the reference system are calculated with emission data for the respective area, using the same characterization factors as for the LCIA. Normalization data is usually included in LCIA methods, and the CML method which is basically adopted in this study provides normalization data

for different geographical and temporal scopes (Netherlands 1997, Western Europe 1995, World 1995).

However, the scope of the normalization data should correspond to the scope of the study if possible. The present work investigates different sanitation systems which are operated in Germany, using technology that is available for the year 2007. Hence, normalization should be carried out in relation to the latest available emission data for Germany. Consequently, emission data for Germany is collected for the year 2004 (or latest available data) and used for the calculation of specific normalization scores (Table 16):

$$I_{i,GER2004} = \frac{\sum_j C_{ij} \cdot m_{j,GER2004}}{pop_{GER2004}}$$

with $I_{i,GER2004}$ = *normalization score for indicator i in Germany 2004 (unit: inhabitant equivalents [pe*a])*

C_{ij} = *characterization factor for flow j and indicator i*

$m_{j,\ GER2004}$ = *total flow j in Germany 2004 (or latest available data)*

$pop_{GER2004}$ = *population of Germany 2004 (82.523.000 inhab.)*

The emission data used for the calculation of normalization scores for Germany 2004 eventually suffers from incompleteness due to the lack of emission data. For some indicators (CED, GWP, AP, EP), the most relevant emission data is readily available. For toxicity indicators, emissions to air are available for heavy metals and some organic and inorganic pollutants. Data for emissions to surface waters and soil are limited to heavy metals.

If emission data for the calculation of normalization scores is inconsistent, it can lead to a biased normalization (Heijungs et al., 2007). The normalized indicator results are then calculated either too high or too low, and their use in further interpretation can be misleading. It is recommended to detect possible biases in normalization scores (e.g. by a thorough analysis of the contributing substances) and discuss their influence on the interpretation. Biased normalization seems to be particularly important for impact categories where many substances are relevant, i.e. any form of toxicity assessment and resource depletion (Heijungs et al., 2007).

Table 16: Normalization data

Indicator		Germany 2004	Western Europe 1995
		[1]	*[2]*
Cumulative energy demand	*MJ/(pe*a)*	168468	-
Abiotic depletion potential	*kg Sb-eq/(pe*a)*	32.6*	32.6
Global warming potential	*kg CO_2-eq/(pe*a)*	12202	14600
Acidification potential	*kg SO_2-eq/(pe*a)*	13.5	84.2**
Eutrophication potential	*kg PO_4-eq/(pe*a)*	6.5	38.4**
Human toxicity potential	*kg DCB-eq/(pe*a)*	7266	23300
Freshwater aquatic ecotoxicity potential	*kg DCB-eq/(pe*a)*	88.9	1550
Terrestrial ecotoxicity potential	*kg DCB-eq/(pe*a)*	70.1	146
Terrestrial eutrophication	*m^2 UES/(pe*a)*	1011.3	-
Aquatic eutrophication of inland waters	*kg NO_3-eq/(pe*a)*	37.1	-
Aquatic ecotoxicity	*kg TEG-eq/(pe*a)*	1722063	-
Terrestrial ecotoxicity	*kg TEG-eq/(pe*a)*	1969880	-

DCB: 1,4-dichlorobenzene, TEG: triethylene glycol, UES: unprotected ecosystem
** ADP adopted from CML method due to lack of data*
*** calculated with baseline characterization factors (fate and exposure excluded)*
Sources:
[1] see annex 12.3
[2] CML method (Guinée et al., 2002)

3.8.5 Grouping and weighting

Grouping and weighting the different indicator results of Life Cycle Impact Assessment is another optional step in LCA to come to conclusive statements in an environmental comparison (ISO 14040, 2006). In comparing two or more systems, the environmental pros and cons usually have to be grouped and weighted against each other to identify the preferable system in terms of environmental impacts. This so-called valuation of the results is both based on scientific findings and certain standards of value. Thus, it combines scientific findings with subjective value judgements, and the application of

certain criteria for valuation of LCIA results has to be addressed in a transparent and reproducible manner to comply with the ISO standard.

Different schools of valuation exist for grouping and weighting of LCIA results in the scientific community. Some methods try to reduce the comparison to just one representative "overall score" of the systems or products under investigation (e.g. Ecoindicator 99), weighting the different endpoint categories (e.g. human health, ecosystem quality etc) to add up the numeric results to one final score. These approaches are traditionally targeting LCA for product development, so that the designer can easily choose the preferable product with lower environmental impact without further knowledge of the LCA background.
Other valuation methods are based on ranking of the various impact categories, i.e. allocating a higher priority to a specific impact category than to another. The method applied in this work has been developed by the German Federal Environmental Agency (UBA method, Schmitz and Paulini, 1999) and is based on a subjective ranking of the indicators following a certain methodology.

In detail, the UBA method is designed in alignment with the superior protected assets of environmental policy, i.e. human health, the structure and functionality of ecosystems, and natural resources. Thus, it assigns different priorities for each indicator on a relative ordinal scale (A = highest priority to E = lowest priority) based on three different criteria:

1. **Ecological hazard**: how serious the potential hazard for the protected asset is for the respective impact category, in particular concerning potential effects of a damage, reversibility, spatial extent, and uncertainties of the cause-effect-chain.
2. **Distance to target**: how distant the existing environmental status is to the status of ecological sustainability, i.e. regarding the distance to quantifiable environmental quality goals, the extent of necessary reduction, actual emission trends (rising, stable, lowering), and the feasibility of abatement measures.
3. **Specific contribution**: how large the normalized impact score is in relation to the total impact in Germany.

Whereas the first two criteria are derived by subjective valuation, the third criterion is determined simply by computation, relating the various normalized indicator results to the score of the indicator with highest normalized score (for calculation details see Schmitz and Paulini, 1999). Finally, the evaluations of all three criteria are combined to formulate a final statement about the ecological priority of the respective impact category.

Based on the proposal of the Federal Environmental Agency (UBA) for the ranking of impact categories (Schmitz and Paulini, 1999), priorities for all impact categories are derived in an ordinal scale (Table 17). It has to be noted that choices for ecological hazard and distance to target are adopted from UBA except for the toxicity indicators (HTP, FAETP, TETP) which are explicitly not included in the UBA proposal. For a comprehensive valuation of all indicators, it has been decided to include toxicity indicators here (= "modified UBA method"). For the ecological hazard, level A is attributed for all three toxicity indicators. The distance to target is estimated to level C for all three indicators, based on the relatively good state of environment that is the result of environmental policy in Germany for the last 50 years. The resulting ecological priority of each indicator is defined based on a specific scheme. In the valuation of this study, all indicators are attributed either medium or large priority (Table 17). To reveal the influence of the modification of the UBA valuation method, the original UBA ranking (without toxicity indicators) is applied in sensitivity analysis.
For a comparison between two scenarios, the relative indicator results are illustrated in a T-diagram. Now, relative scores with the same priority can be counterbalanced against each other. If this comparison leads to a definitive advantage of one scenario over the other, the result can be seen as significant. Otherwise, the differences between the scenarios should be described as "insignificant".

Table 17: Ranking of LCIA indicators based on ecological hazard, distance to target, and specific contribution

	Ecological hazard	Distance to target	Specific contribution	Resulting ecological priority
CED	C	B	D	**Medium**
GWP	A	A	E	**Large**
ADP	C	B	D	**Medium**
EP	B	C	A	**Large**
AP	B	B	B	**Large**
HTP	A	C	E	**Medium**
FAETP	A	C	A	**Large**
TETP	A	C	A	**Large**

A = highest priority, E = lowest priority
Ranking based on UBA method (Schmitz and Paulini, 1999) extended with toxicity indicators

3.9 Interpretation and sensitivity analysis

This last step of an LCA brings together the results of classification, characterization, normalization and grouping or weighting to an overall interpretation. Significant issues influencing the result of the LCA study should be addressed in this part. The interpretation of the results should explicitly take into account the

- goal and scope of the study
- data quality of the Life Cycle Inventory
- significant parameters of LCI
- methodological limitations
- the influence of value-choices (e.g. in weighting)
- results of sensitivity analysis.

Completeness, sensitivity and consistency have to be checked to assess the reliability of the results. For a comprehensive LCA, the uncertainties connected to the results of the study should be clearly communicated to support the significance of the study (ISO 14044, 2006). Finally, conclusions and recommendations are presented together with limitations of the study.

Sensitivity analysis

Sensitivity analysis is a systematic procedure for the estimation of the influence of variations in LCI data or LCIA methodology on the final results of the LCA. In this study, two types of sensitivity analysis are implemented:

A) sensitivity analysis of LCI data
B) sensitivity analysis of LCA methodology

Sensitivity analysis of LCI data

In this part, specific parameters of the inventory (e.g. energy demand of sub-processes, transport distances, process layout, flow characteristics) are varied over a defined range to observe the resulting effect on single LCIA indicators. Thus, the robustness of the LCA results and the related conclusions can be tested to:

- quantify the influence of boundary conditions, assumptions, or other uncertainties for specific LCI data
- identify relevant processes which should be further optimized
- identify relevant key parameters for the ecological assessment to facilitate data acquisition for future LCA studies in this field.

Sometimes, the variation of a system parameter can lead to necessary adjustments in the respective scenario or in other scenarios. If e.g. improved urine separation efficiency leads to more urine available for fertilizing, this results in a) increased working time and emissions of the tractor and b) more production and application of mineral fertilizer in other scenarios. All necessary adjustments are fully implemented while calculating the indicator results of sensitivity analysis.

Sensitivity analysis of LCA methodology

This part of sensitivity analysis relates to methodological choices and their influence on the results and conclusions of this study. For certain impact categories (eutrophication, ecotoxicity), alternative indicators are calculated to reveal a possible influence of the indicator choice on the results. Other methodological issues are e.g. related to functional definitions inside the inventory data (e.g. effluent concentrations) or modification of the valuation method. Sensitivity analysis of LCA methodology can be helpful to identify important definitions or choices that have been made prior to the setup of the substance flow model (LCI) and their influence on the outcomes. Thus, it can be revealed to what extent these definitions can determine the LCA results, and the respective findings can be seen in relation to these issues.

4 Life Cycle Inventory

This chapter contains a detailed description of the Life Cycle Inventory, i.e. the input data that is used to model the relevant system components of the different scenarios and related subsystems. Core processes of wastewater collection and treatment ("system operation") are described with a high level of detail, explicitly stating process layout, underlying assumptions, data origin, and parameter adjustment. Data for system construction (infrastructure) is generated by a simplified layout of the systems in combination with secondary data from a real city area. Data for background processes such as energy supply and transport is adopted from existing databases and is described briefly. The resulting substance flow models for all scenarios are implemented in UMBERTO® software (IFU and IFEU, 2005).

4.1 System operation

This part describes the Life Cycle Inventories for the operation of the different processes in each scenario. It includes the following processes:

A) Conventional sanitation system

- drainage
- activated sludge plant with sludge stabilisation
- composting of biowaste

B) Source-separation systems

- urine separation, storage and treatment
- gravity drainage and composting of faeces with biowaste
- vacuum drainage and digestion of faeces with biowaste
- vacuum drainage and digestion of faeces and urine with biowaste
- greywater treatment in
  - activated sludge plant (sequencing batch reactor)
  - planted soil filter
  - membrane bioreactor with non-potable reuse

C) Application of mineral and organic fertilizers

The different scenarios of conventional and source-separation systems are built by different combinations of the processes listed above. An exact definition of each scenario and its relevant processes is given in chapter 3.5.

4.1.1 Conventional sanitation system

Scenario R

The activated sludge plant is operated with extended nutrient removal, i.e. denitrification and chemical P elimination. Excess sludge is stabilised by anaerobic digestion, dewatered and co-incinerated with municipal waste. The energy demand of wastewater treatment is minimized by using energy-efficient aggregates and surplus energy from sewage gas combustion (Figure 21).

Scenario R_{min}

Wastewater treatment in this scenario is operated according to minimum standards for wastewater discharge, i.e. without extended nutrient removal. The activated sludge plant eliminates organic matter and ammonia (nitrification). The excess sludge is stabilised simultaneously by extended aeration (Figure 22). Stabilised sludge is co-incinerated in a waste incineration plant.

Scenario R_{agri}

This scenario resembles the second reference scenario (R) in terms of wastewater treatment (extended nutrient removal, sludge digestion, optimized energy demand). Here, the stabilised sludge is directly applied in agriculture as organic fertilizer (Figure 21).

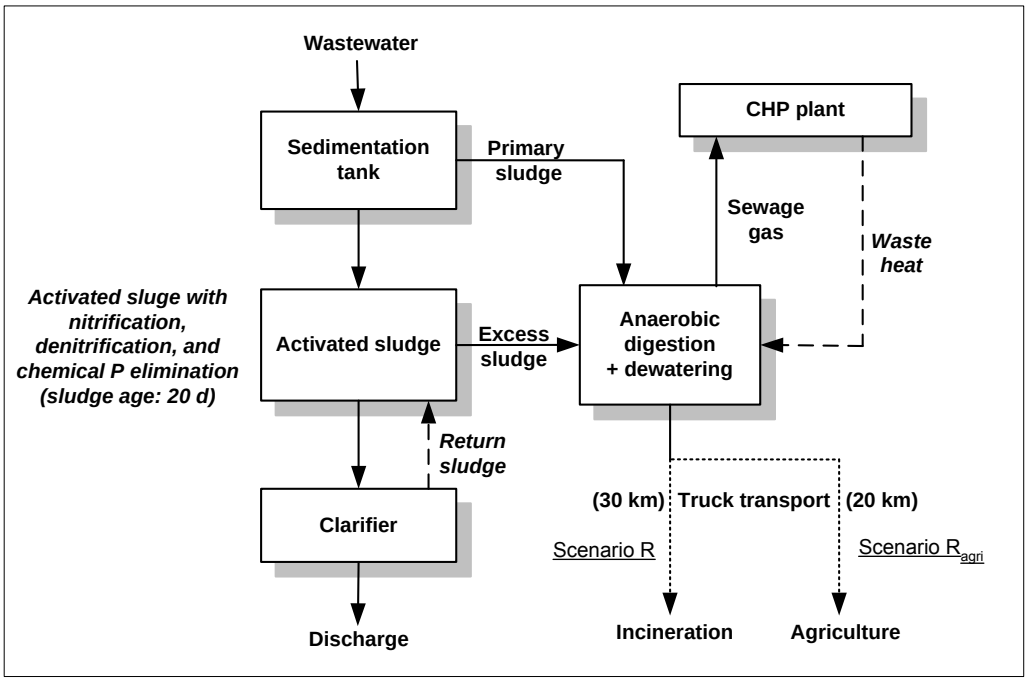

Figure 21: System layout of conventional wastewater treatment with anaerobic sludge stabilisation (R and R_{agri})

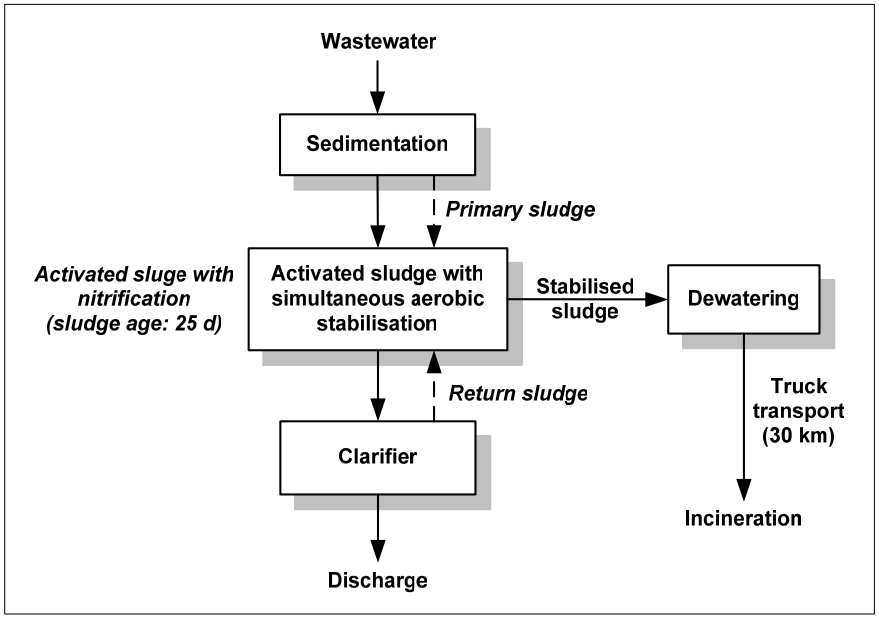

Figure 22: System layout for conventional wastewater treatment with aerobic sludge stabilisation (R_{min})

4.1.1.1 Drainage

The drainage of mixed wastewater in the reference scenarios is done by gravity. The operation of lifting or pumping stations in the sewer system and the associated energy demand is not considered here. Even though terrain properties often require lifting or pumping of wastewater (e.g. in the Berlin sewer network), gravity drainage per se does not need energy for operation. For reasons of simplicity, the investigated area is assumed to be plain, so that pumping is not necessary in the conventional system.

Possible transformations of wastewater components during drainage (e.g. the formation of H_2S or NH_3) are neglected. The influence of infiltrating groundwater or possible leakages of wastewater pipes are not considered as well.

4.1.1.2 Activated sludge plant

For the operation of the conventional wastewater treatment plant, an LCA process model is applied that has been developed at TU Berlin. It describes the operation of an activated sludge plant and covers the processes of mechanical, biological and (optional) chemical treatment of conventional wastewater, including the stabilisation of sewage sludge.

The model calculates the allocation of the different elemental components (C, N, P etc) of influent wastewater to the output flows of effluent, air, and sewage sludge. It does not provide a dynamic load-dependent 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 (Table 18). A detailed description of the process model is provided in annex 12.4. Here, the most important process parameters are briefly discussed:

- *Sedimentation:* Particulate fractions of COD (35% of influent), N (11%), and P (17%) are removed via sedimentation and do not contribute to aeration energy or chemical demand.
- *Elimination of organic matter:* Chemical oxygen demand (COD) is eliminated to 95% in all scenarios.
- *Nitrification/denitrification:* Removal of dissolved N is specifically targeted only in scenarios R and R_{agri} (90% removal of dissolved N). Dissolved N removal in scenario R_{min} (50%) is due to nitrogen demand for bacterial growth and partially due to unintended denitrification processes (e.g. in clarifier).
- *Chemical P elimination:* Dissolved P is chemically precipitated by ferric salt addition in scenarios R and R_{agri} (β factor = 1.5). In the minimum scenario R_{min}, P is removed only by microbial uptake for biomass growth. Rates of biological P removal are estimated according to common German rules for the design of activated sludge plants (ATV, 2000).
- *Heavy metals:* Heavy metals are bound in element-specific fractions to sewage sludge.
- *Sludge age:* The sludge age is estimated to 25 days in case of simultaneous aerobic sludge stabilisation (= extended aeration). For anaerobic sludge stabilisation, a sludge age of 20 days is sufficient to achieve denitrification (ATV, 2000). The stabilised sludge is dewatered by a mobile centrifuge to 40% dry matter content. The sludge liquor (including significant loads of NH_4) is recycled to the influent.
- *Sludge yield:* The amount of excess sludge is determined by the yield coefficient of 0.67 g C_{sludge}/g $C_{eliminated}$ (ATV, 2000).

Energy demand

The demand for electric energy is calculated via the particular energy requirements of the relevant processes (e.g. aeration, mixing, pumping, sludge treatment etc) (Table 19). The resulting total energy demand (23.4 kWh/(pe*a) for scenarios R and R_{agri} and 27.8 kWh/(pe*a) for scenario R_{min}) is in the lower range of average values for German WWTPs (Table 20), reflecting modern process design and machinery. The volume-related energy demand is substantially higher than the average of German WWTPs due to the treatment of concentrated wastewater without stormwater.

In scenarios with optimized energy demand (R and R_{agri}), the net energy balance is further improved by energy production from sewage gas combustion. The sewage gas is

combusted in a combined heat and power plant (electrical/thermal efficiency: 32%/57%). The thermal energy demand for digester heating is met by the waste heat from the CHP plant.

Table 18: Process parameters of conventional WWTP

Parameter		Scenario $R + R_{agri}$	Scenario R_{min}	Remarks
Removal in sedimentation				
COD	[%]	35	35	Particulate fraction of influent load
N	[%]	11	11	Particulate fraction of influent load
P	[%]	17	17	Particulate fraction of influent load
Removal in biological stage				
COD	[%]	95	95	Total incl. $COD_{particulate}$
NH_4	[%]	> 93	> 93	Nitrification
$N_{dissolved}$	[%]	90	50	Denitrification
$P_{dissolved}$	[%]	95	40	Total elimination
by chemicals	[%]	60	-	Addition of Fe (β = 1.5)
Heavy metals	[%]	60 – 85	60 – 85	Depending on element
Sludge treatment				
Stabilisation		anaerobic	aerobic	
Sludge age	[d]	20	25	ATV, 2000
Sludge yield Y	[g C_{sludge} / g C_{elim}]	0.67	0.67	ATV, 2000
Dry matter content of stabilised sludge	[%]	40	40	Dewatering with centrifuge, sludge liquor recycled to influent
Energy demand				
Electric	[kWh/m³]	0.61*	0.65	Total (incl. sludge stabilisation and dewatering)
	[kWh/pe*a]	23.4*	27.8	Total (incl. sludge stabilisation and dewatering)
Thermal	[MJ/m³]	0	-	Digester heating by heat of CHP plant

** without benefit from biogas use (12.1 kWh/pe*a)*

Table 19: Energy demand for WWTP in reference scenarios

Process	Scenarios R + R_{agri}	Scenario R_{min}
	[kWh/pe*a]	[kWh/pe*a]
Pumping + primary treatment	2.6	2.5
Aeration*	18.3	22.8
Benefit from denitrification	- 5.1	- 2.8
Mixing	1.9	3.2
Sludge treatment	4	0.9
Other	1.7	1.2
Total	**23.4**	**27.8**
Benefit from sewage gas usage	-12.1	0
Net energy demand	**11.3**	**27.8**

Calculated values (LfU, 1998; Müller et al., 1999), see annex 12.4.4 for details
** Aeration energy required for carbon removal and nitrification*

Table 20: Comparison of energy demand for wastewater treatment with average values of municipal WWTPs in Germany

Energy demand	This study		Average values of Germany*		
	Scenarios R + R_{agri}	Scenario R_{min}	Median	80%-percentile	weighted average
kWh/(design-pe*a)	23.4	27.8	27	41	24.3
kWh/(pe * a)			41.5	64	31.7
kWh/m³	0.61	0.65	0.32	0.56	0.32

** values from 1097 municipal wastewater treatment plants (LfU, 1998)*

Transfer coefficients

For the wastewater treatment process, exemplary transfer coefficients for each scenario are calculated with the default parameters. They describe the allocation of the elemental input flows to the different output flows for all three reference scenarios (Table 21). Resulting effluent loads and effluent concentrations of wastewater treatment are calculated in chapter 5.1.4 for each reference scenario.

The stabilised and dewatered excess sludge contains 25% of influent carbon, 18% of influent nitrogen, and 50% or 96% of influent phosphorus depending on whether chemical P elimination is applied. These potentially valuable nutrients are recycled to agriculture in scenario R_{agri}, whereas they are lost for recycling if sewage sludge is incinerated (scenarios R and R_{min}).

Table 21: Transfer coefficients of elemental flows in conventional wastewater treatment

Input	Emissions	Scenarios R + R_{agri}				Scenario R_{min}		
		Effluent	*Air*	*Sludge*	*Sewage gas*	*Effluent*	*Air*	*Sludge*
COD-C	COD-C	5.0				5.0		
	HCO_3-C	1.8				1.7		
	CO_2-C		44.7		8.0		68.7	
	CH_4-C				15.4			
	$C_{org, sludge}$			25.1				24.6
N_{total}	NH_4-N	0.9				4.5		
	NO_3-N	6.2				31.1		
	N org	1.8				8.9		
	N_2-N		72.7		0.1		37.5	
	NH_3-N		0.3				0.3	
	N_2O-N		0.4				0.2	
	N_{sludge}			17.6				17.5
P_{total}	P-species	4.2		95.8		49.8		50.2
K	K	95		5		95		5
Heavy metals								
Pb, Cr, Hg		20		80		20		80
Cd		30		70		30		70
Cu		15		85		15		85
Ni		40		60		40		60
Zn		25		75		25		75

Transfer coefficients calculated with LCA model for wastewater treatment (TU Berlin)

4.1.1.3 Collection and composting of biowaste

The kitchen and garden biowaste from households is collected by truck in a stop-and-go mode (7.5 km) and transported to a composting plant (20 km). Possible emissions during storage and collection of biowaste are neglected.

The process layout of the composting plant consists of a pretreatment of biowaste, followed by an encapsulated intensive composting phase in boxes (average retention time: 11 days) and a final stabilisation in open composting for eight weeks (Figure 23). The product is stabilised compost which can be stored without further treatment before direct application in agriculture. Exhaust air from the intensive composting process can contain harmful and malodorous gases and is therefore cleaned in a biofilter. Excess leachate from intensive composting is discharged to the wastewater treatment plant.

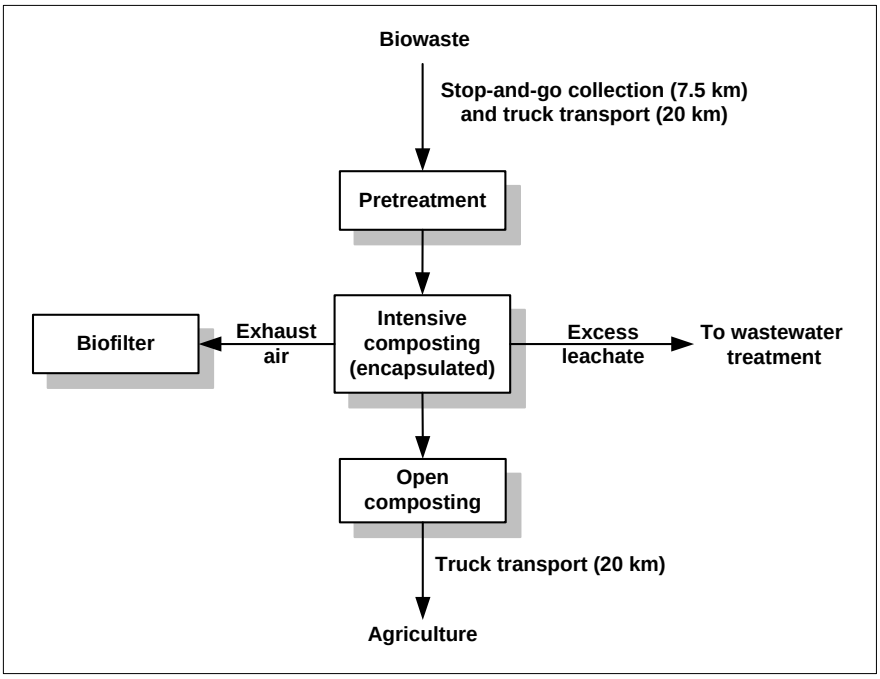

Figure 23: System layout for composting of biowaste

Process inventory

Inventory data of the composting process is basically adopted from a previous LCA study of biowaste composting (Vogt et al., 2002). The pretreatment of biowaste includes chopping and elimination of metals and plastics and requires a considerable portion of the total energy demand (Table 22). The intensive composting is done with active aeration to provide enough oxygen for the aerobic process. During the final open composting stage, the composted material is turned over biweekly with a special turning machine powered by a diesel engine.

Table 22: Energy demand for the composting of biowaste

Process		Energy demand
Pretreatment	[kWh/Mg biowaste]	14.2
Intensive composting	[kWh/Mg biowaste]	10.0
Open composting	[L diesel/Mg biowaste]	0.5

Source: Vogt et al., 2002

Leachate of the intensive composting process is recycled for the moistening of the piles during open composting. Excess leachate (~ 60 L/Mg biowaste) is discharged into the sewer and treated in conventional wastewater treatment. This leachate is usually heavily loaded with organic matter (COD = 20000 – 100000 mg/L (Bidlingmaier, 2000)) and nitrogen. Transfer of phosphorus and salts (K, Cl, heavy metals, etc) into the leachate is neglected.

The final product of the composting process is a stabilized organic fertilizer with a dry matter content of 60%. It can be stored and transported to farms by truck prior to application as a slow-release fertilizer and soil conditioner.

Transfer coefficients

Transfer coefficients of the composting process are calculated for the most important elemental flows (Table 23). A detailed allocation of elemental flows, energy demand, and emissions is given in chapter 4.1.2.2 for the composting of faeces and biowaste, which is described with the same process model.

In total, it is assumed that 60% of the organic carbon and 40% of the nitrogen of biowaste is converted into off-gases or leachate during the composting process. Consequently, gaseous emissions of nitrogen (NH_3, NO_x, and N_2O) and methane (CH_4) are most relevant for the environmental evaluation.

Table 23: Transfer coefficients of biowaste composting

Input	Emission as	Compost	Air	Excess leachate
		[%]	[%]	[%]
Organic carbon	C_{org} in compost	40.6		
	TOC		1.2	
	CO_2-C		55.8	
	CH_4-C		1.8	
	COD-C			0.6
N_{total}	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_{total}			0.3
P_{total}	P in compost	100		
K	K in compost	100		

Source: Vogt et al., 2002

4.1.2 Separation systems

4.1.2.1 Urine separation, storage and treatment

Undiluted urine is separately collected by using separation toilets and drained by gravity to holding tanks (Figure 24). Here it is collected biweekly with tanker trucks and transported (5 km) to a treatment unit, where micropollutants are oxidized by ozone. Treated urine is stored for at least 6 months for hygienisation, before it is transported to farms (20 km) and applied as organic fertilizer in agriculture.

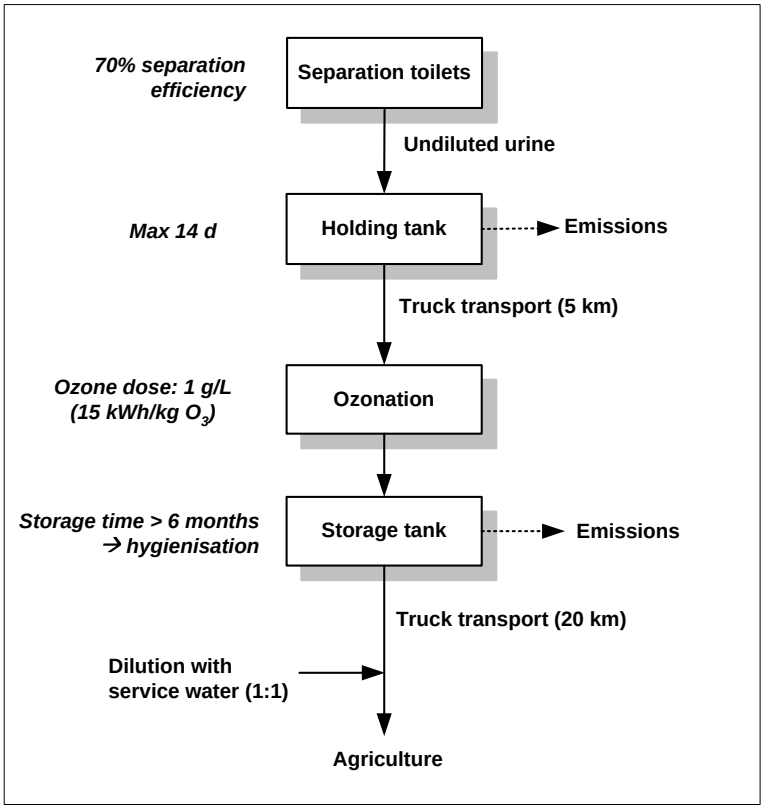

Figure 24: System layout of urine separation and treatment

Separation efficiency

Urine can be separated from remaining toilet wastewater by separation toilets, which are available from various manufacturers (SwedEnviro, 2001). These toilets have two separate outlets: one for urine in the front part and one for faeces and toilet paper in the rear part of the bowl. If the urine outlet is fitted with a movable plug, the collection of undiluted urine without flush water is possible (Roediger, 2007). However, due to incorrect use of the toilets and imperfect design of the bowl, it is not possible to collect 100% of the daily urine of the inhabitants. Experiences from Swedish pilot plants indicate that 60 – 90% of the total urine flow can be separated depending on the motivation of the tenants (Jönsson, 2001). Results from a German pilot plant suggest a separation efficiency of only 33 – 41% (Peter-Fröhlich et al., 2007), but this is mainly attributed to user behaviour and construction deficiencies of the system. In a Swiss pilot project, 70-75% of the expected urine quantity could be recovered with separation toilets (Rossi et al., 2009).

In this study, it is assumed that 70% of the daily urine can be collected without flush water, while 30% of the urine is drained together with faeces and flush water. This seems to be a reasonable mean value which should be achievable with optimized toilet systems and user education. The influence of separation efficiency on the overall results is further investigated in sensitivity analysis.

Urine treatment: micropollutants

Human urine can contain considerable amounts of pharmaceuticals (Lienert et al., 2007; Winker et al., 2008) and hormones, and the fate of these micropollutants during agricultural application of source-separated urine is not properly understood (Winker, 2009). In a Swedish pilot project, untreated human urine is applied as fertilizer after long-time storage, and micropollutants are assessed to pose only minor risks to consumers or the environment (Stockholm Vatten, 2000). However, the need for more research in this field is clearly identified in the Swedish report. In Germany, legal regulations for fertilizers are strict (DüMV, 2003). The approval of new types of organic fertilizers requires evidence for their harmlessness for the environment and the consumers. Following the precautionary principle, human urine should be treated prior to its application as a fertilizer to reach inactivation of pharmaceuticals and hormones.

For the treatment of source-separated urine, a wide range of technical options is available to reach different purposes which could be hygienisation, stabilisation, volume reduction, recovery of nutrients, or handling of micropollutants (Maurer et al., 2006). Treatment options include storage, evaporation, stripping, biological processes, precipitation, ion exchange, membrane filtration, and ozonation.

In this study, the main goal of urine treatment is the inactivation of micropollutants to eliminate possible risks of urine application. Pilot studies have shown the ability of ozonation to inactivate and metabolize pharmaceuticals in human urine (Escher et al., 2006; Fitzke and Geissen, 2007), and ozone is a well-known oxidant in water treatment. Consequently, the present work includes an ozonation step for the oxidation of micropollutants in urine prior to its application in agriculture. Pilot experiments for the ozonation of human urine have determined a suitable ozone dose of 0.6 – 1.3 g O_3/L to oxidize a set of micropollutants below the level of detection (Escher et al., 2006; Dodd et al., 2008). Based on these results, it is assumed that an ozone dose of 1 g O_3/L is adequate to achieve a sufficient inactivation of potentially harmful micropollutants. According to comparable units used in water treatment, an overall energy demand of 15 kWh per kg ozone is estimated for this process (ITT, 2007). Process emissions of ozonation (e.g. via waste air) are neglected.

Urine treatment: volume reduction

Volume reduction may be another target of interest, but the energy demand of suitable treatment technologies can be high and may offset the advantages of small transport volumes (Maurer et al., 2003). Many treatment options still have to be optimized and tested in full-scale to generate reliable data of process performance and energy demand. A Suisse pilot project recently tested the feasibility of a combination of electrodialysis and ozonation for the production of a concentrated fertilizer without micropollutants from source-separated urine (Pronk et al., 2007; Dodd et al., 2008). It should therefore be emphasized here that other options for urine treatment will probably be available in

the near future to facilitate the recovery of urine-bound nutrients in agriculture. Whether the implementation of these technologies has a significant impact on the LCA results, this has to be checked when reliable process data is available.

Storage and application

After ozonation, 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). Cross contamination of separated urine with faecal microorganisms cannot be completely avoided (Hoglund et al., 1998 ; Schonning et al., 2002) and sufficient storage at reasonable temperature leads to the die-off of pathogenic bacteria (Höglund, 2001). After that storing period, the urine is considered to be stabilised and hygienically safe for application as fertilizer. The urine is mixed to stir up possible precipitates and transported to the farms (20 km). Prior to the application, it is diluted with service water (1:1) to minimize potential losses of ammonia during application. The energy demand for the various pumping processes is estimated to 0.08 kWh/m³ urine. The treatment and supply of service water for dilution is assumed to consume 0.1 kWh/m³.

Atmospheric emissions during collection, storage, and application

During the collection and storage of urine, the urea content is hydrolysed to NH_3, and pH rises from 6 to 9. A part of the nitrogen is lost via atmospheric emissions during collection, transport and storage. Estimated transfer coefficients can be found in literature (Table 24).

NH_3 losses in pipe networks (0.01%) and storage tanks (0.003%) are relatively small. Most of the NH_3 evaporates during the application of urine on the fields. Depending on the application technique and weather conditions, the losses of nitrogen via NH_3 evaporation can amount to 2 – 10 % (Johansson et al., 2001). Pilot-scale field tests conducted in Germany resulted in a nitrogen loss of 3 – 10% (Peter-Fröhlich et al., 2007; Muskolus, 2008). In a conservative approach, this study assumes an average loss of 10% of the nitrogen through ammonia evaporation. If suitable application techniques (trailing hoses, injection etc (Döhler, 2001)) are applied, NH_3 volatilisation can presumably be reduced. Losses via N_2O and NO_x are estimated to be equivalent to mineral fertilizer application (EMEP/CORINAIR, 2004.

Table 24: Nitrogen losses during urine collection, storage and application

Losses		Pipes and holding tanks	Storage tanks	Application in agriculture	Source
NH_3-N	[% N in urine]	<1 / **0.01** / <0.3	0.1 / **0.003** / –	2 - 10 / 6 / 1 - 10	1 / 2 / 4
			– / 5 / –	6 / 5 / 3 - **10**	3 / 5 / 7
N_2O-N	[% N in urine]			1.25 **/ 1.25** / 0.3	5 / 6 / 7
NO_x-N	[% N in urine]			0.1 / **0.7**	5 / 6

Bold: this study
Sources:
1) Johansson et al., 2001
2) Vinneras et al., 1998
3) Simons and Clemens, 2004
4) Palm et al., 2002
5) Tidaker, 2003
6) EMEP/CORINAIR, 2004 (equivalent to mineral fertilizers)
7) Peter-Fröhlich et al., 2007; Muskolus, 2008

4.1.2.2 Gravity drainage and composting of faeces with biowaste

For the composting of faeces with biowaste, faeces are collected in standard toilets (preferably with low flush water volume, i.e. 6 L per flush) and drained in a gravity sewer. Flush water is separated from the solids by a solid-liquid separation process to obtain faecal matter with sufficient dry matter content for the aerobic composting process. Solid faecal matter is transported to a composting plant, where it is mixed with pretreated biowaste. The composting plant is a two-stage process: an intensive composting in boxes is followed by an open composting stage for stabilisation (Figure 25). The final product is stabilised compost with high organic matter content, which can be transported to the farms and used in agriculture for soil conditioning.

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 (Gajurel, 2003; Winblad and Simpson-Hébert, 2004; Muskolus, 2008). Laboratory experiments have confirmed that composting with worms ("vermicomposting") seems to be a promising process to convert faeces into an odourless, earth-like material within three months (Shalabi, 2006).

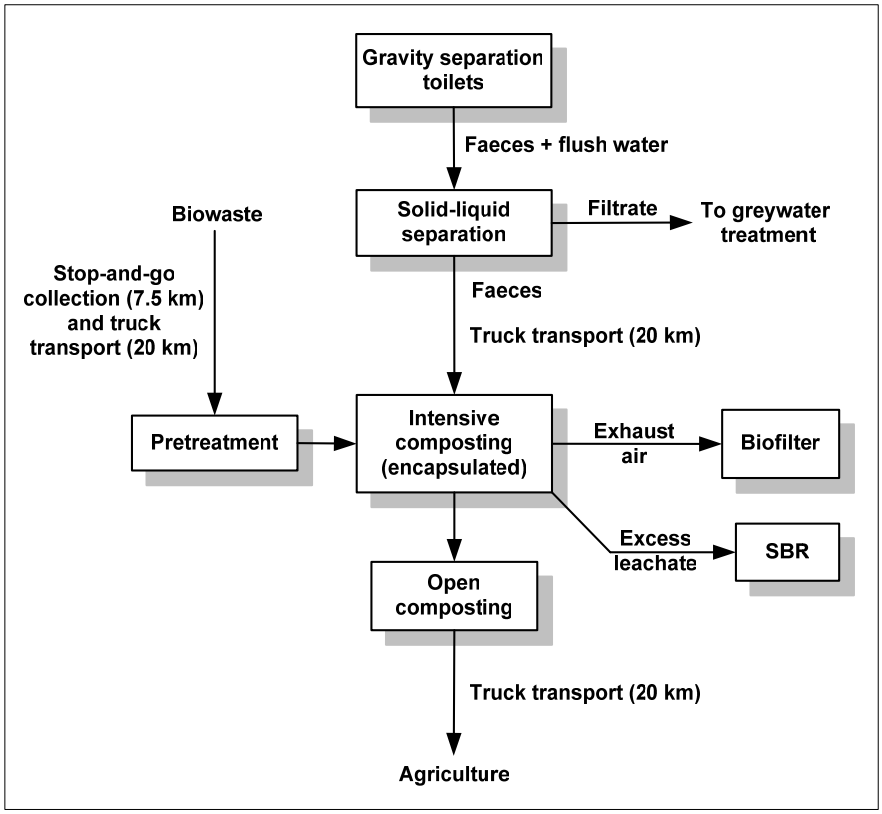

Figure 25: System layout for composting of faeces and biowaste

However, the composting of human faeces together with biowaste has been realized to date only in single houses or small scale units, mostly in a rural context. Its implementation in a high-tech urban environment requires fully automated processes for solid-liquid separation, storage and transport, and large-scale composting of faeces. This may pose difficulties for process engineering in terms of efficiency and reliability.

In particular, the separation of the flush water from the faecal matter without major losses of nutrients and organic matter is a difficult task. First field trials of a separation process based on filter bags are unsatisfactory for large-scale implementation due to insufficient separation efficiency and impractical handling (Peter-Fröhlich et al., 2007). A novel rotating disc filter with ceramic membranes is currently developed at the Fraunhofer Institute in Stuttgart which may allow the continuous operation of a filtration process for blackwater (IGB, 2007). Another approach uses a simple whirlpool surface tension separator ("Aquatron") to separate flush water from faecal matter inside the house (Vinneras, 2004). However, none of these processes has so far been successfully applied in larger scale. For this study, the process data for solid-liquid separation is estimated from dewatering and thickening devices of sewage sludge treatment.

Similarly, there is only few data on the material flows during faeces composting. To overcome this lack of data, it is decided that the inventory of this process should be based on inventory data of biowaste composting (Vogt et al., 2002). Although the process configuration of biowaste composting may be different from faeces composting, the data is thought to provide a good estimation of resource demand and emissions of the composting process. More research on large-scale composting of faeces is required to accurately predict energy demand and emissions as well as the allocation of the elemental flows on the different output fractions (compost, exhaust air, and leachate).

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 obtained. The demand of electric energy and chemicals is estimated from sewage sludge treatment (Table 25). Residual sludge is temporarily stored and transported to the composting plant by truck (20 km). Possible anaerobic decomposition of the faeces sludge during storage is neglected.

Table 25: Energy and chemical demand of solid-liquid separation

Input	Amount	Remarks
Polyacrylamide	10 g/kg dry matter	adapted from the thickening of sewage sludge (Müller et al., 1994; Schumann et al., 1997)
Electric energy	0.03 kWh/kg dry matter	
Electric energy	0.13 kWh/m³	pumping

Filtrate

The separated flush water ("faeces filtrate") is heavily loaded with water-soluble nutrients and organic matter from faeces and also from urine which is not properly separated in the separation toilets (Gajurel et al., 2003). In this study, faeces filtrate is pumped to the greywater sewer and treated in greywater treatment.

The quality of the filtrate is decisive for the amount of nutrients which are lost for recycling purposes and have to be removed in greywater treatment. The filtrate composition is assumed based on data from a pilot plant for faeces dewatering in bags (Peter-Fröhlich et al., 2007), faeces dewatering in specific composting systems ("Rottebehälter",Gajurel et al., 2003) and consideration of input flows (Table 26):

- Organic matter mainly originates from dissolving of faecal matter
- N and P loads originate from misdirected urine

- Salts and heavy metals: concentrations in filtrate are calculated as average concentrations of urine and flush water (i.e. it is assumed that faeces-derived salts and heavy metals do not dissolve in the flush water)

In case of greywater treatment in MBR and non-potable reuse (scenario SC3), the faeces filtrate which is treated together with greywater is partially used for toilet flushing, thus creating a closed loop in the model which cannot be simulated by the UMBERTO® software. For this scenario, the quality of the faeces filtrate is approximated by an iterative calculation.

Table 26: Composition of filtrate from faeces dewatering

Substance	Concentration			Load
	[mg/L]			*[kg/(pe*a)]*
	[1]	[2]	This study	This study
Volume				8780
Total organic carbon	398-758*	1023*	200	1.8
Nitrogen (total)	17-152	141	121	1.1
Phosphorus	1-35	16	12	0.1
Salts, heavy metals	average concentration of urine and flush water			

** COD*
1) Gajurel, 2003 (Rottebehaelter)
2) Peter-Fröhlich et al., 2007 (mean values with filter bags)

Pretreatment of biowaste

Biowaste has to be conditioned and homogenised prior to the composting process. Possibly hazardous material for the compost quality has to be removed (e.g. plastics, metals). Biowaste with high water content has to be chopped and mixed with structure material if necessary (e.g. saw dust) 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 well as the further handling of the removed plastics and metal. The energy demand for the pretreatment steps (Table 27) includes an exhaust air treatment.

Table 27: Energy demand for pretreatment of biowaste

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

Source: Vogt et al., 2002

Intensive composting

The automated process of intensive composting is conducted in closed boxes with heat insulation over a period of 11 days. The respective allocation coefficients of the different elemental components of biowaste to the output fractions (Table 28) are adopted from biowaste composting and are related to an 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. In total, 16% of carbon and 10.6% of nitrogen are lost with exhaust air or leachate.

Leachate

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 here (Gronauer et al., 1997). Data for heavy metal concentration in composting leachate indicates that the proportion of heavy metals lost with the leachate is small (Fricke, 1990). It is assumed that excess leachate is only charged with nutrients and organic matter (COD > 10000 mg/L). This leachate is treated in a separate activated sludge plant (sequencing batch reactor) next to the composting plant. The process model for the activated sludge plant is similar to that for greywater treatment (see chapter 4.1.2.5) with nitrification, denitrification, and chemical P elimination. Due to the high COD load of the leachate, a total energy demand of 2 kWh/m³ is assumed for this process.

Table 28: Substance flows of intensive composting of faeces and biowaste

Input	Output	Source
Water content	In compost: 50% water content	1
	In biofilter: difference	
	In leachate: 125 L/Mg biowaste input	1
Organic carbon	In compost: $C_{out} = (1 – 0.16) * C_{in}$	1
	In biofilter: CO_2-C_{out} = 0.14 * C_{in} CH_4-C_{out} = 0.004 * C_{in} NMVOC-C_{out} = 0.003 * C_{in}	1, 3
	In leachate: TOC = 0.013 * C_{in}	1
Nitrogen	In compost: N_{out} = (1 – 0.106) * N_{in}	1
	In biofilter: NH_3-N_{out} = 0.096 * N_{in} N_2O-N_{out} = 0.002 * N_{in} N_2-N_{out} = 0.002 * N_{in}	1
	In leachate: NH_4-N_{out} = 0.003 * N_{in} N org. $_{out}$ = 0.003 * N_{in}	1
Phosphorus	In compost: P_{out} = 0.997 * P_{in} In leachate: P_{out} = 0.003 * P_{in}	assumption: P_{out} = 20 mg/L
Salts and heavy metals	Accumulation in compost, fraction in leachate is neglected	1, 2
Inert substances	Remain in compost	
Energy demand (incl. biofilter)	10 kWh/Mg waste	1

Sources:
1) Vogt et al. 2002 (for biowaste composting in closed boxes, retention time : 11 days)
2) Gronauer et al. 1997
3) Leinemann 1998 (similar values from tunnel reactors)

Exhaust air

In most composting plants, the exhaust air is cleaned in a biofilter to prevent the output of malodorous gases and ammonia. Allocation coefficients of the biofilter are compiled from different sources (Table 29). It is assumed that 60% of ammonia is partially converted to N_2O and NO in the biofilter (Cuhls, 2001; Clemens and Cuhls, 2003). Data for nitrogenous air emissions from a biofilter varies considerably in literature. The amount of N_2O emissions is likely to depend on the ammonia load to the filter.

Table 29: Allocation coefficients of biofilter

Input to biofilter	Output		Source
H_2O-vapor	H_2O_{out}	$= H_2O_{in}$	
CO_2-C	CO_2-C_{out}	$= CO_2$-C_{in}	1, 2
CH_4-C	CH_4-C_{out}	$= CH_4$-C_{in}	1
NMVOC-C	NMVOC-C_{out}	= 0.5 * NMVOC-C_{in}	2
NH_3-N	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}	1,3
N_2O-N	N_2O-N_{out2}	$= N_2O$-N_{in}	1,2,3
N_2-N	N_2-N_{out}	$= N_2$-N_{in}	2

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

Open composting

After the intensive degradation stage in boxes, the composted mix of faeces and biowaste is stabilised in open piles for 8 weeks to obtain a marketable product. A mechanical circulation of the piles with appropriate machinery is necessary biweekly to ensure oxygen supply and prevent anaerobic decomposition. The allocation coefficients for open composting (Table 30) are basically adopted from biowaste composting (Vogt et al., 2002). During the stabilisation phase, 52% of organic carbon and 33% of nitrogen are converted to gaseous emissions. No leachate is produced in this stage, because excess moisture evaporates directly to the atmosphere. The final product is stabilised compost with 40% water content which can be stored or directly transported to farms for application in agriculture.

Faeces can be contaminated with pathogenic microorganisms. These pathogens have to be reliably inactivated in the composting process to prevent any potential hazard for farmers and agricultural products. Usually, the long stabilisation phase leads to a substantial reduction in pathogens due to elevated temperatures and microbial competition. In this study, it is assumed that pathogens are eliminated in open composting so that no further hygienisation of the final compost is necessary.

Table 30: Substance flows of open composting of pre-composted faeces and biowaste

Input	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-C_{out} = 0.494 * C_{in} CH_4-C_{out} = 0.016 * C_{in} NMVOC-C_{out} = 0.010 * C_{in}	1
Nitrogen	In compost: N_{out} = (1 – 0.33) * N_{in}	1
	In off-gas: NH_3-N_{out} = 0.317 * N_{in} N_2O-N_{out} = 0.007 * N_{in} N_2-N_{out} = 0.007 * N_{in}	1
Phosphorus, salts, heavy metals and inert substances	Accumulation in compost, no leachate	1,2
Pathogenic microorganisms	Reliable inactivation of pathogens	
Diesel fuel (circulation)	0.76 L/Mg waste	1

Sources:
1) Vogt et al. 2002 (open composting in piles, retention time: 56 days)
2) Gronauer et al. 1997

4.1.2.3 Vacuum drainage and digestion of faeces with biowaste

For the digestion of faeces with biowaste, faeces are drained in a vacuum system with very small amounts of flush water. From the vacuum station, the mixture is pumped directly to the biogas plant where it is mixed with pretreated biowaste. Faeces can be heavily contaminated with pathogenic microorganisms, so the mixture has to pass a hygienisation stage. Pathogens are reliably inactivated by pasteurization of the substrate (70°C for 1 h). The hygienisation step is usually applied ahead of the digester to minimize the risk of recontamination (ATV, 1996). In the digester, the organic content is partially degraded under anaerobic conditions and transformed into biogas (mainly CO_2 and CH_4) which is used in a combined heat and power plant (CHP plant) to generate electricity and heat. The remaining residual sludge still contains valuable nutrients and some organic material, and it can be used as fertilizer in agriculture after stabilisation (Figure 26). Hence, this option of faeces treatment uses the energetic content of the faecal organic matter without major losses of nutrients N and P while delivering both energy and fertilizer (Wendland et al., 2007).

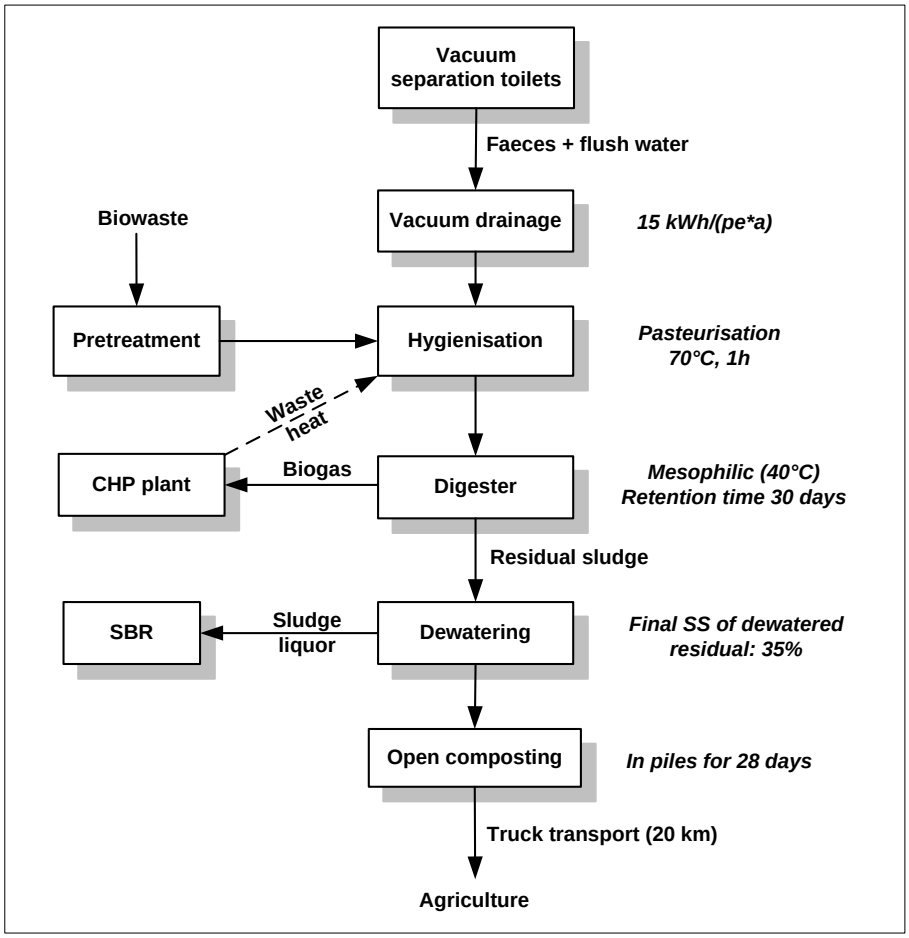

Figure 26: System layout for vacuum drainage and co-digestion of faeces with biowaste

Vacuum drainage

The vacuum system for the drainage of faeces can operate with very small amounts of flush water (1.2 L/flush). A vacuum pump imposes a slight negative pressure (ca. -0.5 bar) on the pipe network, sucking the mixture of faeces and flush water towards the vacuum tank. Interface valves and small interim holding tanks inside the houses allow for a comfortable operation without inconvenience for the customer. From the main vacuum station, faeces are pumped in pressure lines to the biogas plant. The energy demand for the vacuum system can vary considerably depending on the size and layout of the system and the wastewater volume (Table 31). For this study, the energy demand for the vacuum pumps is estimated to 15 kWh/(pe*a). Pressure pumps deliver the faeces mixture to the biogas plant (0.05 kWh/m³).

Table 31: Energy demand of vacuum systems

System	Amount of wastewater	Size	Energy demand	Remarks	Source
	*[L/(pe*d)]*	*[pe]*	*[kWh/(pe*a)]*		
Flintenbreite	5	108	51	Not working to capacity	1
			17	Possible	1
Vauban	8.4	40	7		2
Hannover	9	80	27	Annual period	3
			9	Possible	3
Stahnsdorf	9	9	3.1	Calculated	4
ATV	150	--	36	Combined wastewater	5
This study	**5.8**	**5000**	**15**		

Sources:
1) Otterwasser, 2005
2) Schneidmadl, 1999 (calculated from operating time)
3) Herrmann and Hesse, 2002
4) Peter-Fröhlich et al., 2007 (calculated for optimized system)
5) ATV, 1995

Pretreatment

An automated rake system protects the digester from interfering objects in faeces sludge. Biowaste is shredded in smaller parts, and plastics and metals are eliminated in a wet separation process ("swim-and-sink-process") combined with a pulper to support the hydrolysis of the biowaste. The energy demand of the different processing steps is

adopted from biowaste digestion. After the pretreatment and mixing of the substrates, the water content is adjusted to 90 – 97% with service water if necessary.

Table 32: Energy demand for the pretreatment of digester substrate

Process	Electric energy demand	Source
Rake	0.07 kWh/m³	MURL, 1999
Shredder	1.2 kWh/Mg (1)	Vogt et al., 2002
Exhaust air treatment	8.1 kWh/Mg	Vogt et al., 2002
Wet processing / pulper	150 kWh/Mg	Vogt et al., 2002

1) related to wet mass of waste
2) related to dry matter

Hygienisation

The hygienisation of the substrate is ensured by a pasteurisation step prior to the digester. Pathogenic microorganisms are inactivated by means of elevated temperature (70°C for one hour). The waste heat from the CHP plant is used for hygienisation. A heat exchanger reclaims a major part of the thermal energy from the hot effluent of the hygienisation unit. Process parameters are estimated from sewage sludge hygienisation (Table 33). If the waste heat of biogas combustion is not sufficient for the hygienisation process, extra fuel can be added to the CHP plant.

Table 33: Parameters for the thermal energy balance of hygienisation

Parameter			Remarks
Specific heat capacity of dry matter	1.05	*MJ/(Mg*K)*	MURL, 1999
Specific heat capacity of water	4.19	*MJ/(Mg*K)*	
Hygienisation temperature	70	*°C*	Retention time: 1h
Starting temperature of substrate (annual mean)	15	*°C*	Assumption
Substrate temperature after heat exchanger	35	*°C*	
Proportion of the recovered waste heat from output	85	*%*	Assumption
Energy losses through transmission	5	*%*	According to sludge digestion (MURL, 1999)

Digester

During the anaerobic fermentation process, the organic matter is dissolved in water (hydrolysis) and converted by microorganisms 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 of the substrate is between 90 and 97%. Digesters can be operated at different temperatures, however most of the digesters in Germany are operated in mesophilic conditions (30 – 42°C). Typically, retention times are around 30 – 40 days, in which around 50% of the degradable substrate is converted into biogas (BLU, 2004; Stadtmüller, 2004).

This study assumes a mesophilic digestion process (40°C) with an average retention time of 30 days. Co-digestion of faeces and biowaste is a reliable process under these conditions and has been tested successfully in various pilot studies (Otterwasser, 2005; Kujawa-Roeleveld et al., 2006; Wendland et al., 2007). However, the process has not yet been realized in full-scale. Consequently, the operating conditions and the energy demand have to be estimated from literature (Table 34). It is assumed that the operating temperature of the digestion process can be maintained without heating. The process itself generates thermal energy from microbial activity and the digester substrate is delivered with a temperature of 35°C from hygienisation.

Table 34: Parameters of the mesophilic digestion process

Parameter		Unit
Digester		
Digester temperature	40	*°C*
Retention time (Digester and stabilisation)	30	*days*
Water content of digester substrate	95	*%*
Electric energy demand of digester	3	*kWh/Mg fresh mass*
Thermal energy demand of digester	0*	
Dewatering of residual		
Water content of dewatered residual	65	*%*
Dosage of coagulation aid	4	*kg/Mg dry matter*
Water demand for solving of coagulation aid	200	*kg/kg*
Electric energy demand for dewatering	30	*kWh/Mg dry matter*

** energy transfer from hygienisation is sufficient to maintain operating temperature*
Source: Vogt et al. 2002

The expected methane yield of the relevant substrates (faeces, misdirected urine, and biowaste) is estimated from pilot studies and literature (Table 35). Methane yield is calculated based on the content of organic dry matter of the different substrates, assuming a certain amount of generated biogas with a specific methane content (ρ_{CH4}= 0.72 kg/m³).

After the digestion process, 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 combusted in the CHP plant. Prior to composting, the stabilised digester residual is dewatered to 65% dry matter content to establish aerobic conditions during the subsequent composting process.

Table 35: Biogas and methane yield of different substrates in relation to input mass to digester

Substrate	Organic dry matter (oDM)	Biogas	Methane	Methane	Source
	[% dry matter]	*[m³/kg oDM]*	*[m³/kg oDM]*	*[kg/kg oDM]*	
Faeces	93	0.45	0.29	0.21	1
Urine*	75	0.34	0.22	0.16	2
Kitchen biowaste	72	0.45	0.28	0.20	3
Garden biowaste	71	0.48	0.30	0.21	4

** misdirected urine which is not properly separated in toilets*
Sources:
1) Estimation according to Otterwasser, 2005; assumed methane content in biogas 65%: Kujawa-Roeleveld et al., 2003 report a methane content of 70% for blackwater digestion
2) Data from Otterwasser, 2005 suggests 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 (BLU, 2004)
4) Estimation according to mean values for loppings (BLU, 2004)

Dewatering of digester residual

Dewatering of digester residual is done mechanically with a centrifuge and coagulation aids. Sludge liquor from digester residual can contain considerable amounts of organics and nutrients (Loll, 1999). Here, transfer ratios are adopted from biowaste digestion (Vogt et al., 2002). Concentrations of N_{total} and P are assumed to be higher than in the literature due to the contribution of misdirected urine (Table 36). The elution of heavy metals is estimated according to the dewatering of sewage sludge.

Sludge liquor has to be treated in an activated sludge plant (SBR) with nutrient elimination prior to discharge. Process data for the SBR plant is basically adopted from the SBR model for greywater treatment (see 4.1.2.5). Elimination ratios are set to 95% for TOC, 97% for NH_4-N, 85% for N_{total}, and 95% for P to reach legally requested effluent concentrations. Due to the high load of organics and NH_4-N, the energy demand for the treatment of sludge liquor is relatively high (0.68 kWh/m³).

Table 36: Transfer ratios from digester residual to sludge liquor during dewatering

Substance	Transfer ratio [%]	Source/Remarks
C_{org}	2	Vogt et al., 2002
N_{total}	35*	85% as NH_4-N, 15% as N_{org}
P	15*	
K	50	Vogt et al., 2002
Mg, Ca	10	Vogt et al., 2002
Heavy metals		
Pb, Cd, Cr, Hg	20	estimation according to dewatering of sewage sludge
Cu	15	
Ni	40	
Zn	25	

** elevated due to misdirected urine*

Open composting

The dewatered digester residual is further stabilized in open composting in piles. Due to lack of appropriate data, the relevant substance flows are based on an LCA model for composting of residual from biowaste digestion (Vogt et al., 2002). 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. Leachate containing organic carbon and nitrogen is treated together with sludge liquor. Losses of phosphorus, salts and heavy metals with the leachate are neglected.

Table 37: Substance flows of open composting of digester residual

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-C_{out} = 0.290 * C_{in} CH_4-C_{out} = 0.01 * C_{in} NMVOC-C_{out} = 0.006 * C_{in} In leachate: TOC_{out} = 0.004 * C_{in}	1
Nitrogen	In compost: N_{out} = 0.83 * N_{in}	1
	In off-gas: NH_3-N_{out} = 0.163 * N_{in} N_2O-N_{out} = 0.003 * N_{in} N_2-N_{out} = 0.003 * N_{in} In leachate: NH_4-N_{out} = 0.0005 * N_{in} N org_{out} = 0.0005 * N_{in}	1
Phosphorus, salts, heavy metals and inert substances	Assumption: accumulation in compost	1, 2
Diesel fuel (circulation)	2.2 L/Mg waste	1

Sources:
1) Vogt et al. 2002 (open composting of digester residual, retention time: 28 days)
2) Gronauer et al. 1997

Combined heat and power plant

The biogas is combusted in a CHP plant to generate electric and thermal energy. A potentially required gas conditioning prior to the combustion process is neglected. A part of the generated biogas has to be flared in case of system malfunction or storage overflow. It is assumed that 5% of the total sewage gas volume is flared (Ronchetti et al., 2002), generating emissions which are comparable to the combustion in the CHP plant. A small proportion of the sewage gas (0.75%) is lost by accidental leakage (Ronchetti et al., 2002), causing respective emissions of methane.

The CHP plant is equipped with a spark-ignition engine in lean combustion mode with high excess air, so that legal air emission standards can be met easily. CHP parameters and emission factors (Table 38) are compiled from an LCA study (Ronchetti et al., 2002) and the Umberto® database (IFU and IFEU, 2005)

Table 38: Parameters and emission factors for CHP plant

	This study	Biogas-CHP	Natural gas-CHP
Source		Ronchetti et al., 2002	Umberto® database (IFU and IFEU, 2005)
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
Emissions *in mg/MJ*			
CH_4, combustion	2,5	2,48	3,78
CO_2	*	81.308	55.151
NO_x (as NO_2)	38	37,85	62,98
N_2O	1,6	--	1,57
CO	51	50,93	51,17
SO_x (as SO_2)	30	29,91	0,43
NMVOC	2,5	2,48	4,72
Dust	1,6	--	1,57

** depending on input (CO_2 + CH_4) minus CO*

4.1.2.4 Vacuum drainage and digestion of faeces and urine with biowaste

In scenarios using vacuum toilets without urine separation, the mixture of faeces and urine is drained by a vacuum system and co-digested with biowaste. Basically, this process is comparable to the digestion of faeces and biowaste described in chapter 4.1.2.3. Vacuum system, pretreatment of digester substrate, hygienisation and digestion are modelled using the same data sets and assumptions.

The digester residual contains a large amount of dissolved nutrients due to the contribution of urine. These valuable nutrients would be lost to a great extent for fertilizing purposes if the residual sludge would be dewatered and composted. Hence, it is assumed that the digester residual is directly applied as fertilizer, comparable to liquid animal manure (Figure 27). Although the transport volume is relatively large and NH_3 emissions during application are presumably higher than for composted residual, the direct application of digester residual seems to be the best solution for optimum utilization of the nutrients if urine is not separated in the toilets.

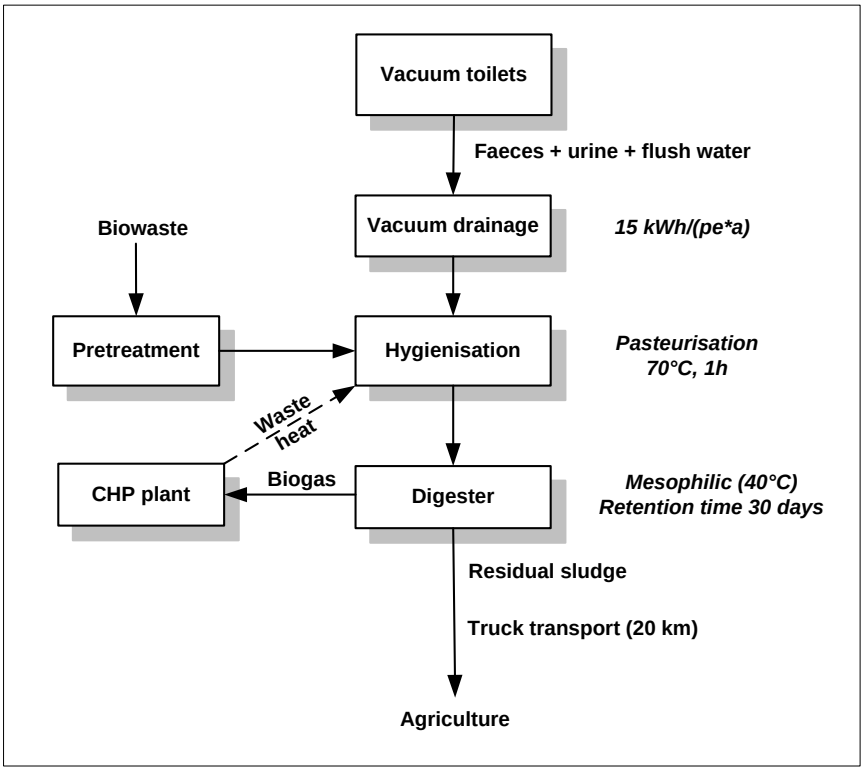

Figure 27: System layout for vacuum drainage and co-digestion of faeces and urine with biowaste

If digester residual is directly applied in agriculture, no emissions from sludge liquor treatment and open composting arise. The complete process is encapsulated without any atmospheric emissions (except from biogas combustion in CHP plant). Possible

emissions during storage of digester residual are difficult to quantify and are neglected here. However, the emissions during application of liquid digester residual in agriculture are significantly higher than for dewatered and stabilised residual.

4.1.2.5 Sequencing batch reactor for greywater treatment

Greywater can easily be treated in a conventional activated sludge process due to its low load of organic matter and nutrients (Figure 28). Particulate matter is separated in an upstream sedimentation tank. The activated sludge process for greywater treatment is designed as a sequencing batch reactor (SBR) providing carbon removal, nitrification, and chemical P elimination. All stages of the process proceed in a single reactor within a temporal sequence. If nitrogen load to greywater treatment is increased by the co-treatment of faeces filtrate (scenario SC2), a denitrification phase (mixing and no aeration) has to be implemented to comply with legal effluent standards. Excess sludge is withdrawn and stabilised aerobically together with primary sludge from sedimentation. Digestion of the greywater sludge for energy recovery is not considered here due to the low load of organic matter in the sludge. Stabilised sludge is dewatered and incinerated.

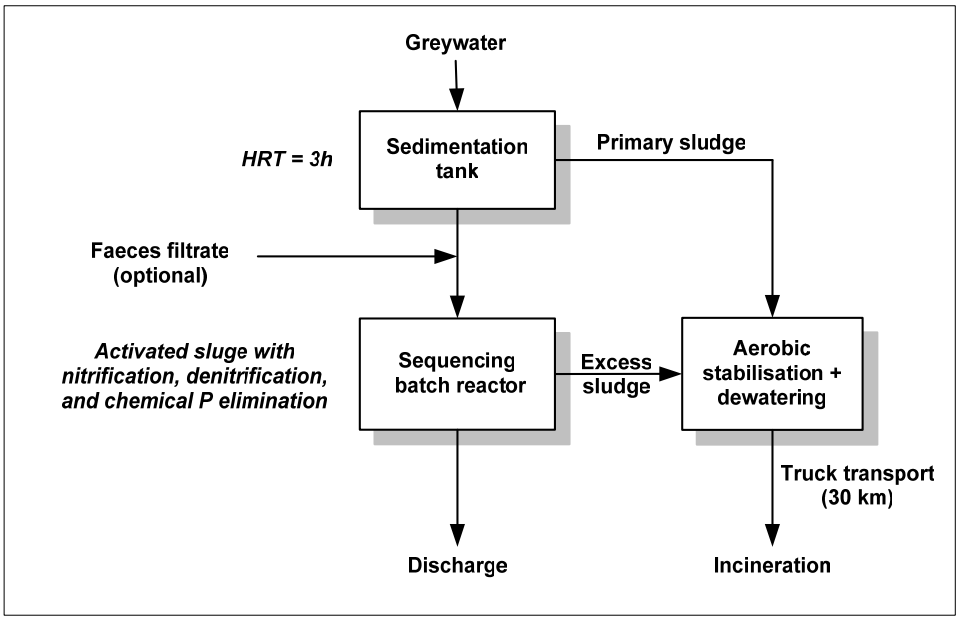

Figure 28: System layout for greywater treatment in sequencing batch reactor

The LCA process model is basically adapted from conventional wastewater treatment (see chapter 4.1.1.2). However, the different composition of greywater compared to combined wastewater requires some adjustments (Table 39):

- Particulate matter (15% of TOC, 11% of N_{total}, 10% of P_{total}) is separated in sedimentation, and faeces filtrate is optionally added after this stage
- The distribution of nitrogen and phosphorus to the different output flows is determined with the same calculatory approach as for the conventional WWTP, but with different elimination ratios.
- For the carbon balance, excess sludge production is estimated via a constant factor (0.6 g dry matter per g TOC eliminated) for an average sludge age of 25 days.
- The influence of recycling of sludge liquor to the influent is neglected here. Sludge liquor is directly discharged to surface waters.

Table 39: Process parameters of SBR for greywater treatment

Parameter			Remarks
Removal in sedimentation			
TOC	*[%]*	15	particulate fractions of influent loads
N	*[%]*	11	
P	*[%]*	10	
Removal in biological stage			
$TOC_{dissolved}$	*[%]*	93	reflecting typical removal efficiencies of C and N in SBR plants (Helmreich et al., 2000; Steinmetz et al., 2002)
NH_4-N	*[%]*	96	
$N_{dissolved}$	*[%]*	70	
$P_{dissolved}$	*[%]*	86	with 20% biological P uptake
Heavy metals	*[%]*	60 – 85	depending on element
Inorganic salts	*[%]*	0	neglected
Sludge age	*[d]*	25	for aerobic sludge stabilisation (ATV, 1997)
Sludge production	*[g dry m./g TOC_{el}]*	0.6	assumption: 70% of dry matter are organic matter, which has 50% carbon content
Energy demand	*[kWh/(pe*a)]*	14.5*	without faeces filtrate
	*[kWh/(pe*a)]*	16.1*	with faeces filtrate

** calculated according to LCA model of conventional WWTP (except energy demand for aerobic stabilisation: 1.8 kWh/kg $C_{org,degraded}$)*

Elimination of nitrogen and phosphorus

The removal ratios for N and P are lower in greywater treatment (70 and 86%) than for combined wastewater in the conventional scenario (90 and 95%), even though both waters are treated with the same technology. This is due to the lower influent N and P loads of greywater, which would lead to unrealistic low effluent concentrations of N and P if the removal ratios of conventional wastewater treatment would be applied. Consequently, N and P removal in greywater treatment is adjusted to smaller total elimination ratios to result in reasonable effluent concentrations. Effluent of greywater treatment still has lower N and P concentrations than conventional wastewater treatment (cf chapter 5.1.4), thus reflecting improved effluent quality of low-loaded greywater treated with conventional activated sludge technology.

Energy demand

The energy demand is calculated from the particular processes (greywater lifting, aeration, mixing, co-precipitation, sludge dewatering, and auxiliaries) according to the LCA model for the conventional activated sludge plant. For aerobic stabilisation, an additional energy demand of 1.8 kWh per kg of eliminated C_{org} is assumed for aeration and mixing (estimated from conventional WWTP with aerobic stabilisation).

Transfer coefficients

The resulting transfer coefficients include the entire process with sedimentation, SBR, sludge stabilisation and dewatering (Table 40). They are calculated here for scenarios without co-treatment of faeces filtrate. The transfer of salts into sludge is neglected here, while the transfer of heavy metals into sludge is calculated with elimination ratios of conventional activated sludge plant (60-85% depending on element).

Table 40: Transfer coefficients of elemental flows in greywater treatment with sequencing batch reactor*

Input	Emission as	Effluent	Air	Sludge
		[%]	*[%]*	*[%]*
TOC	TOC	5.8		
	CO_2-C		78.2	
	C_{org} in sludge			16.0
N_{total}	NH_4-N	2.0		
	NO_3-N	15.5		
	N org	10.2		
	N_2-N		39.5	
	NH_3-N		0.3	
	N_2O-N		0.2	
	N in sludge			32.3
P_{total}	P-species	9.3		90.7
K	K	100		
Heavy metals				
Pb, Cr, Hg		20		80
Cd		30		70
Cu		15		85
Ni		40		60
Zn		75		25

** including sedimentation, SBR, sludge stabilisation and dewatering; without faeces filtrate*

4.1.2.6 Soil filter for greywater treatment

Due to its low load of nutrients and organic matter, greywater can be treated appropriately in a planted soil filter ("constructed wetland"). This natural treatment process is based on the elimination of pollutants during the passage of wastewater through a soil filter that is planted with reed. The root system of the 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. These microorganisms are mainly responsible for the degradation of organic matter and nitrogen, whereas the contribution of the reeds 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). The effluent is characterized by low turbidity, good bacteriological quality, and low carbon and nutrient load. Soil filters are known as a reliable process for wastewater treatment if sufficient surface area is provided to ensure adequate hydraulic loading rates (ATV, 1998; Cooper, 2005).

Process layout

Upstream of the soil filter, particulate matter is removed by sedimentation to reduce the pollutant load on the filter and prevent clogging by particle aggregation on the filter surface. Sludge from sedimentation stage is dewatered and transported to an incineration plant (stabilisation is neglected here due to low sludge volume). The soil filter is operated as a vertical-flow filter with intermittent loading of greywater (Figure 29). Optionally, faeces filtrate can be added to particle-free greywater after sedimentation stage. Long-term experience has shown that no excess sludge is accumulating in the filter (Geller and Thum, 1999). Reed is mowed once each year and added to composting or digestion process.

Process inventory

Particulate matter (15% of TOC, 11% of N, 10% of P) is removed by sedimentation. The subsequent soil filter provides organic carbon removal, nitrification and partial denitrification, and phosphorus removal (Table 41). Nitrogen removal occurs by denitrification processes in anoxic micro-zones of the soil filter and by plant uptake. While carbon and nitrogen are eliminated by microbial activity, phosphorus is mainly eliminated by adsorption on soil particles. Phosphorus elimination can be as high as 90% in fresh soil, especially if the iron content of the filter material is high (Peter-Fröhlich et al., 2007). However, the long-term elimination rate for phosphorus is expected to be around 50% throughout the lifetime of the filter material (Otterwasser, 2005), because it will eventually reach saturation with phosphorus. Adsorption capacity can be extended by the addition of specific adsorption materials into the filter (Arias and Brix, 2005; Molle et al., 2005).

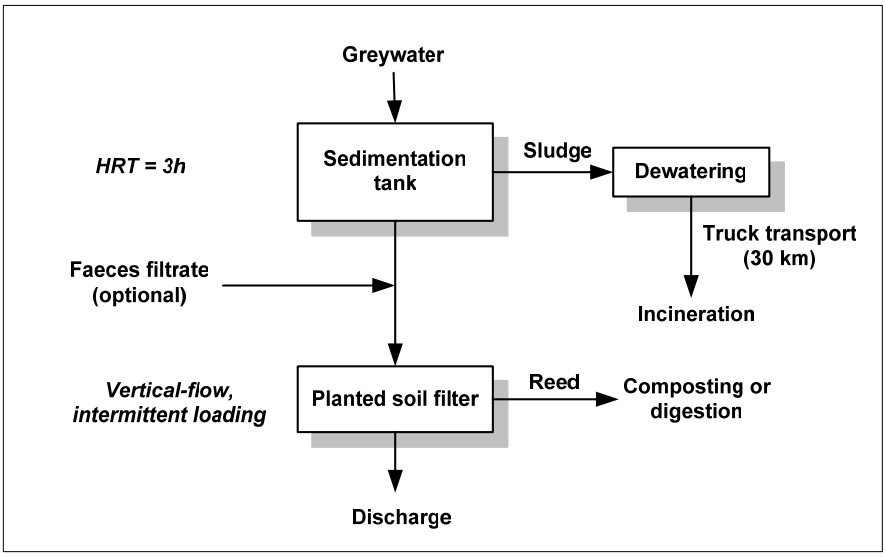

Figure 29: System layout for greywater treatment in soil filter

The removal of inorganic salts (K, S, Ca) occurs only via plant uptake. Heavy metal elimination efficiencies in soil filters have been investigated for Cu and Zn (Vymazal and Krasa, 2003; Otterwasser, 2005). For other heavy metals, elimination has been roughly estimated to be equivalent to conventional wastewater treatment. Their removal in the soil filter occurs mainly by adsorption on soil particles, while plant uptake is negligible.

Energy demand

The energy demand of the soil filter system is determined by the energy consumption of the feed pump. Beside the static pressure head (4 m), a dynamic pressure head of 8 m is calculated for the distribution system (plastic pipes with drilled holes) according to the recommendations of DWA (DWA, 2006).

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 (Geller and Höner, 2003). An average rate of 5 mm/d (= 5 L/m²*d) is assumed in this study.

Surface area

The surface area of a soil filter system has to be adapted to the volume and quality of the influent water. Usually, a surface area of 1 – 3 m² per inhabitant equivalent is sufficient for greywater treatment to prevent clogging of the filter surface, hydraulic overload or oxygen deficiency in the filter (Oldenburg, 2002; Peter-Fröhlich et al., 2007). In this study, the soil filter is designed with a surface area of 2 m² per inhabitant (2.5 m²/pe for treatment of faeces filtrate with greywater).

Table 41: Process parameters of soil filter for greywater treatment

Parameter			Remarks	Sources
Removal in sedimentation				
TOC	*[%]*	15	particulate fraction	
N_{total}	*[%]*	11		
P_{total}	*[%]*	10		
Removal in soil filter				
$TOC_{dissolved}$	*[%]*	90		1, 2, 3
NH_4-N	*[%]*	96		
$N_{dissolved}$	*[%]*	40	partial denitrification	
$P_{dissolved}$	*[%]*	50	adsorption (long-term)	
Heavy metals	*[%]*	60-80	depending on element	see Table 43
Salts	*[%]*		removal by plant uptake	
Energy demand	*[kWh/(pe*a)]*	1.6 (2.1*)	pumping with static and dynamic pressure head	4
Evapotranspiration	*[L/(m²*d)]*	5	1 – 18 mm/d	2, 5
Surface area	*[m²/pe]*	2 (2.5*)	1 – 3 m²/pe	5, 6

** in scenario SC2 due to extra volume of faeces filtrate*
Sources:
1) Bahlo, 1999
2) Langergraber, 2005
3) Otterwasser, 2005
4) DWA, 2006
5) Oldenburg, 2002,
6) Peter-Fröhlich et al., 2007

Plant uptake

The plant uptake of organic carbon, nutrients, and inorganic salts is estimated from literature (Table 42). 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. 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.

Uptake of inorganic salts is roughly estimated. The accumulation of heavy metals in plants is negligible (~ 1%) (Geller and Thum, 1999). 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).

Table 42: Plant uptake of specific elements in soil filter

Substance			Remarks	Sources
Dry matter	*[g/(m²*a)]*	1000	Plant production	Peverly et al., 1995
C_{org}	*[g/(m²*a)]*	370		Peverly et al., 1995
N	*[g/(m²*a)]*	10		Langergraber, 2005
P	*[g/(m²*a)]*	3		Langergraber, 2005
K	*[g/(m²*a)]*	10	Estimated	
S, Ca	*[%]*	1	Estimated	
Cl, Na	*[%]*	0	Estimated	
Heavy metals	*[%]*	1	Estimated	Geller and Thum, 1999

Transfer coefficients

Overall transfer coefficients for sedimentation and soil filter treatment are calculated with the LCA model (Table 43). Compared to the SBR process, the soil filter is characterized by relatively low nutrient removal and higher effluent loads of nitrogen and phosphorus. The advantages of the soil filter are its low energy demand and low sludge production.

Table 43: Transfer coefficients for greywater treatment in soil filter

Input	Emission as	Effluent	Air	Reed	Filter material	Sludge
		[%]	*[%]*	*[%]*	*[%]*	*[%]*
Organic carbon	TOC	8.5				15.0
	CO_2-C		76.5			
N_{total}-N	NH_4-N	2.2				
	NO_3-N	30.7				
	N org	20.5				
	NH_3-N		0.3			
	N_2O-N		0.2			
	N_2-N		30.9			
	N			4.2		11.0
P_{total}-P	P-species	45		3.3	41.7	10
K	K-species	97.3		2.7		
*Heavy metals**						
Pb, Cr, Cu, Hg, Zn		20		1	79	
Cd		30		1	69	
Ni		40		1	59	

Transfer coefficients calculated with LCA model including sedimentation + soil filter, without co-treatment of faeces filtrate
** estimated with transfer coefficients of conventional wastewater treatment except Cu and Zn (Vymazal and Krasa, 2003; Otterwasser, 2005), transfer into reed roughly estimated to 1% of influent load (Geller and Thum, 1999)*

4.1.2.7 Membrane bioreactor for greywater treatment and reuse

If non-potable reuse of purified greywater is targeted (e.g. for toilet flushing), greywater can be treated in an activated sludge process combined with a membrane stage, a membrane bioreactor (MBR). Ultrafiltration membranes are submerged directly into the activated sludge tank to replace the conventional clarification stage for sludge separation. The greywater is drawn through the membranes by a slight negative pressure (outside-in mode), leading to an efficient separation from the activated sludge and an effluent with low turbidity and good hygienic quality. Effluent from MBR processes usually has a sufficient quality for direct reuse as service water (Jefferson et al., 2000). Another advantage of MBR technology is the reduction of required tank volume by a factor of 3 – 4 compared to conventional activated sludge plants (Pinnekamp and Friedrich, 2006). A drawback of this advanced technology is the high energy demand for aeration and the need for periodic cleaning of the membranes with chemicals to ensure constant flow rates and prevent membrane fouling.

Process layout

For the protection of the membrane modules against coarse materials, hair etc, a pretreatment of the greywater with fine sieving is recommended (Frechen et al., 2006). After the pretreatment, faeces filtrate can be added to the process if required. The membrane bioreactor operates with nitrification, denitrification, and chemical P elimination. Sludge is separated by passing the effluent through ultrafiltration membranes. Excess and primary sludge is stabilised via extended aeration, dewatered, and incinerated. The recycling of sludge liquor to the influent is neglected due to the linear model structure. Additionally, a proportion of the effluent is stored in holding tanks to be reused for toilet flushing (Figure 30).

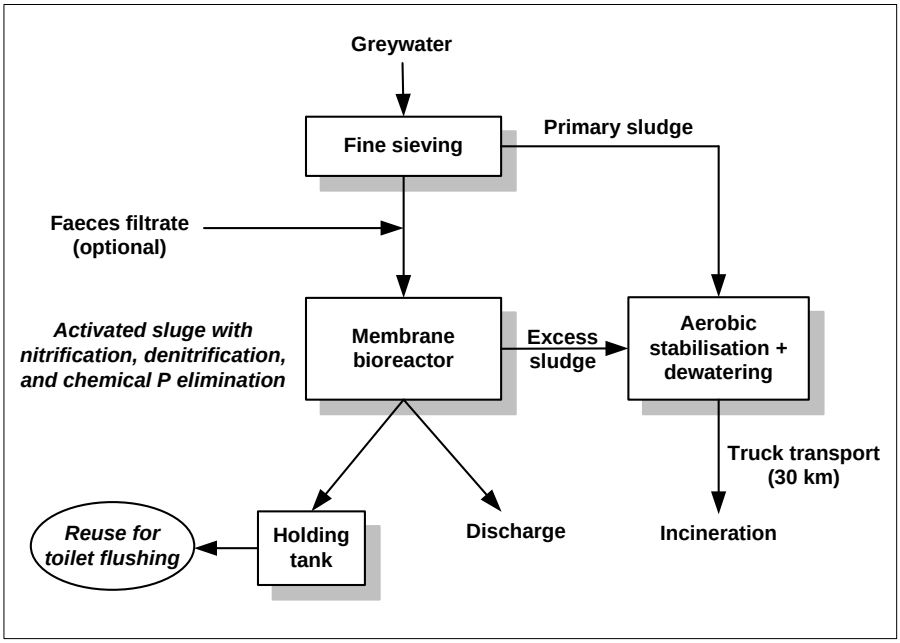

Figure 30: System layout for greywater treatment in membrane bioreactor with partial reuse

Process inventory

In principle, the process model of the MBR is adapted from the conventional SBR plant for greywater treatment (see chapter 4.1.2.5) in terms of emissions and resource demand. Adjustments have been made in process performance (= elimination ratios), sludge production, and energy demand (Table 44):

- Particulate matter (15% of TOC, 11% of N, 10% of P) is separated by fine sieving or inside the MBR without demand for aeration energy or chemicals. Energy demand for fine sieving is assumed to be 0.01 kWh/m³.
- Process performance in terms of carbon oxidation, nitrification, and denitrification is estimated following experiences from an MBR pilot plant treating municipal greywater (Peter-Fröhlich et al., 2007).

- Phosphorus removal via chemical precipitation is assumed to be more efficient (95%) than in the SBR plant (86%), because MBR processes are known to have superior particle retention capacity compared to traditional clarifiers, leading to lower P effluent concentrations (Pinnekamp and Friedrich, 2006).
- Transfer of heavy metals into sludge is estimated according to the SBR process, while transfer of inorganic salts is neglected.
- Excess sludge production is typically lower than in an SBR process. Pilot experiments resulted in an average value of 0.11 g dry matter/g COD_{elim} (~ 0.3 g dry matter/g TOC_{elim}) for a sludge age of 20 d (Peter-Fröhlich et al., 2007).
- Extensive bubble aeration to prevent blockage and fouling of the membrane modules leads to a high energy demand of MBR systems. They have an average energy demand of 0.7 – 1.5 kWh/m³ for treatment of combined wastewater (Pinnekamp and Friedrich, 2006) and 0.5 – 0.9 kWh/m³ for greywater (Peter-Fröhlich et al., 2007).
- Energy and chemical demand for membrane cleaning is neglected here.

Table 44: Process parameters of MBR for greywater treatment

Parameter			Remarks
Elimination in fine sieving			Energy demand: 0.01 kWh/m³
TOC	*[%]*	15	particulate fractions of influent loads
N	*[%]*	11	
P	*[%]*	10	
Elimination in MBR			
$TOC_{dissolved}$	*[%]*	93	estimated from MBR pilot plant for greywater treatment (Peter-Fröhlich et al., 2007)
NH_4-N	*[%]*	96	
$N_{dissolved}$	*[%]*	60	
$P_{dissolved}$	*[%]*	95	with 20% biological P uptake
Heavy metals	*[%]*	60 – 85	depending on element
Inorganic salts	*[%]*	0	neglected
Excess sludge production	*[g dry m./g TOC_{el}]*	0.3	for sludge age of 20 d (Peter-Fröhlich et al., 2007)
Energy demand	*[kWh/m³]*	0.6	0.5 – 0.9 kWh/m³ (Peter-Fröhlich et al., 2007)
	*[kWh/(pe*a)]*	20.2 (25.6*)	incl. pretreatment, aerobic sludge stabilisation (1.8 kWh/kg $C_{degraded}$) and dewatering

** with faeces filtrate (scenario SC3)*

Transfer coefficients

The transfer coefficients for the greywater treatment in an MBR process (Table 45) are calculated for the entire process with fine sieving, MBR, sludge stabilisation and dewatering. Compared to the SBR process, the MBR process is characterized by less organic matter in sludge, higher concentrations of nitrogen in the effluent, and better elimination of phosphorus.

Table 45: Transfer coefficients of elemental flows in greywater treatment with membrane bioreactor*

Input	Emission as	Effluent	Air	Sludge
		[%]	*[%]*	*[%]*
TOC	TOC	6.0		
	CO_2-C		82.4	
	C_{org} in sludge			11.6
N_{total}	NH_4-N	2.0		
	NO_3-N	20.7		
	N org.	13.7		
	N_2-N		30.7	
	NH_3-N		0.3	
	N_2O-N		0.2	
	N in sludge			32.4
P_{total}	P-species	4.6		95.4
K	K	100		
Heavy metals				
Pb, Cr, Hg		20		80
Cd		30		70
Cu		15		85
Ni		40		60
Zn		75		25

** for fine sieving, MBR, sludge stabilisation and dewatering; without faeces filtrate*

Greywater reuse

Greywater which has passed the ultrafiltration modules is assumed to be hygienically safe for non-potable reuse (e.g. toilet flushing) without further disinfection (Lazarova et al., 2003). Hence, a part of the MBR effluent is stored in holding tanks and pumped back to the households in a separate service water pipe, providing a pressure of 5 bar (= 0.23 kWh/m³). The calculated elemental composition of reused greywater is adopted for toilet flush water in the substance flow model. This leads to a recursive model structure in scenario SC3 (greywater → flush water → faeces filtrate → greywater), which is overcome by an iterative calculation of greywater composition.

4.1.3 Fertilizer application

For the inventory of the application of different types of fertilizers in agriculture, several processes have to be considered (Figure 31):

- The operation of the agricultural tractor requires diesel fuel for operation and causes emissions
- Emissions arise from the transformation and evaporation of fertilizer components. Emission factors are specific for each type of fertilizer.
- Fertilizing equivalents of the respective fertilizer are determined by the plant availability of nutrients, which are specific for each type of fertilizer.

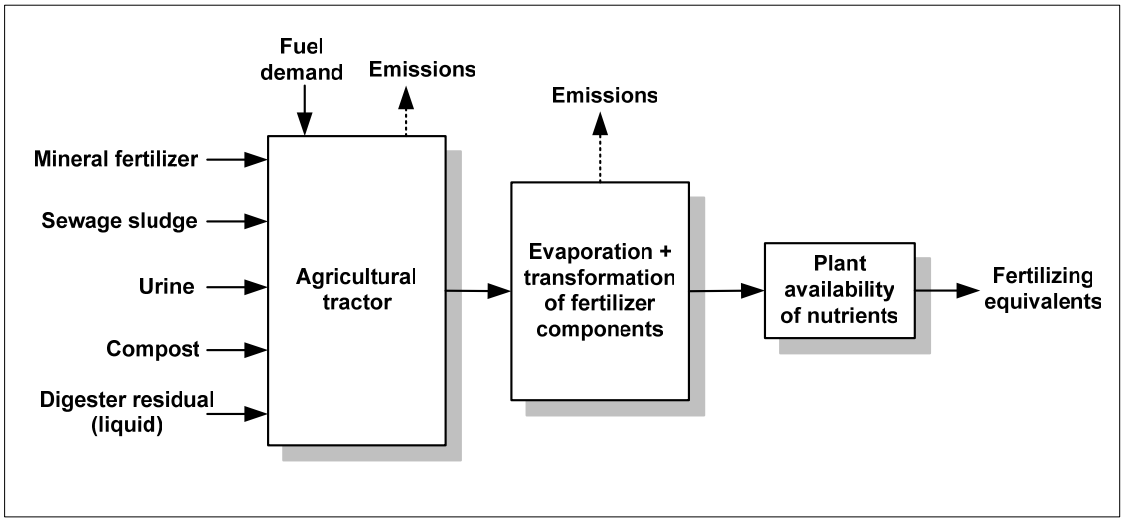

Figure 31: Processes during fertilizer application

4.1.3.1 Agricultural tractor

The inventory for the operation of the agricultural tractor is described with a dataset from Umberto® (IFU and IFEU, 2005). This dataset calculates the energy demand and the emissions from tractor operation depending on engine power, total working time, and engine load levels (Borken et al., 1999):

- Engine load levels for the fertilizer-specific application techniques (solid or liquid) are compiled from a study of agricultural machinery (Rinaldi et al., 2005).
- The total working time for the application of mineral and organic fertilizers is calculated from the field area and the area-related working time (Table 46).

- The field area on which the fertilizers are applied is estimated by assuming a maximum application rate for nitrogen of 110 kg N/(ha*a). For compost, the applicable amount is limited by the total mass (25 t/(ha*a)) due to its low content of plant-available nitrogen. For sewage sludge, the applicable amount is limited to 1.66 t/(ha*a) dry matter by law (AbfKlärV, 1992).

Table 46: Calculation of total working time during fertilizer application

Scenarios	Type of fertilizer	Field area*	Time per area	Regime**	Total working time
		[ha]	*[h/ha]*	*[doses/a]*	*[h/a]*
Reference ($R + R_{min}$)	Mineral	147	0.7	3	308
	Compost	7	1.5 + 2.3	1	11 + 16
Reference (R_{agri})	Mineral	128	0.7	3	270
	Compost	7	1.5 + 2.3	1	11 + 16
	Sewage sludge	51	1.6	2	165
Faeces and urine digestion (V)	Digester residual (liquid)	148	1.6	2	472
Faeces digestion + urine separation (SV)	Mineral	31	0.7	3	65
	Urine	102	1.6	2	328
	Digester residual (stabilised)	15	1.5 + 2.3	1	23 + 35
Faeces composting + urine separation (SC)	Mineral	43	0.7	3	89
	Urine	102	1.6	2	328
	Compost	12	1.5 + 2.3	1	18 + 28

** calculated with maximum amounts: 110 kg N/ha*a or 25 t compost/ha*a or 1.66 t/ha*a sewage sludge (dry matter)*
*** assumed number of doses per year*
Sources: see annex 12.5

4.1.3.2 Volatilisation and transformation of fertilizer components

In general, the gaseous losses of nitrogen during application of the various types of fertilizer heavily depend on the application methods (e.g. spreading, splash plates, trailing hoses, injection, incorporation by ploughing), weather conditions (temperature, wind, precipitation), and soil type and conditions (pH, infiltration capacity, vegetation) (ECETOC, 1994). Consequently, average emission factors in the literature are often given in ranges, depending on boundary conditions. In the field of agricultural LCA, a structured method has been proposed to estimate on-field nitrogen emissions from fertilizer application (Brentrup et al., 2000). It is based on specific parameters for calculating ammonia volatilisation in relation to air temperature, soil infiltration rate, time between application and incorporation, and precipitation. N_2O and NO_x emissions are estimated by a constant factor (Bouwman, 1995). The European emission inventory for agriculture provides similar methods, proposing a simplified or a detailed calculation methodology in relation to the available data (EMEP/CORINAIR, 2004).

However, no specific information about the soil type and further conditions during application are applicable for the present study. Therefore, it is decided to follow a simplified method with constant factors for the calculation of emissions. These figures are adopted from emission inventories and other LCA studies, amended by field data from pilot projects of source-separation systems (Table 47). They can vary significantly under real conditions, but they are expected to be representative for an average loss over a longer period of time.

Mineral fertilizer

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 ammonia emissions can be performed with average emission factors (EMEP/CORINAIR, 2004). Emission factors for six different nitrogen fertilizers are connected to nutrient content and market share to calculate the average NH_3 emission rates. For nitrous oxides, constant factors of 1.25% N_2O-N/kg N (Bouwman, 1995) and 0.7% NO_x-N/kg N (Bouwman et al., 2002) are used. These emissions occur during microbial processes of nitrification and denitrification. The factors are related to the nitrogen input after NH_3 volatilisation, which predominantly occurs earlier than the nitrous oxide emissions (Brentrup et al., 2000). The carbon dioxide emitted from urea hydrolysis (0.59 g CO_2/g N) originates from fossil sources and contributes to climate change.

Table 47: Emission factors for mineral and secondary fertilizers during agricultural application

	NH_3	$N_2O^{\#}$	$NO_x^{\#}$	CO_2 fossil	CO_2 regen.	Source
	g NH_3-N/ g N	*g N_2O-N/ g N*	*g NO_x-N/ g N*	*g CO_2/g N*	*g CO_2-C/g C*	
Mineral N fertilizer	0.05	0.0125	0.007	0.59	0	1,2
Sewage sludge	0.08	0.0125	0.007	0	0	3
Compost from biowaste/faeces	0.05	0.0125	0.007	0	0.5**	4
Urine	0.1	0.0125	0.007	0*	0	5
Digester residual, stabilised	0.063	0.0125	0.007	0	0.5**	4
Digester residual, liquid	0.22	0.0125	0.007	0	0	6

 # emissions from nitrification and denitrification, factor is related to applied N after NH_3 volatilisation
** 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) NH_3 emissions estimated between mineral fertilizer and urine due to lack of data
4) according to compost from biowaste/digester residual in Vogt et al., 2002
5) see chapter 4.1.2.1 for details
6) NH_3 emissions from spreading of cattle slurry (ECETOC, 1994)

Organic fertilizers

The application of organic fertilizers such as sewage sludge, stabilised urine or compost is connected with gaseous nitrogen emissions as well (Table 47). NH_3 emissions of urine application are estimated to 10% of $N_{applied}$ from data of field trials. For liquid digester residual, no data from field trials is available. Due to the mixture of digested faeces and urine, the respective NH_3 emissions during application are estimated in analogy to cattle slurry (22% of $N_{applied}$). For compost, assumptions from an LCA study of biowaste composting are adopted.

Specific emission factors for N_2O and NO_x are not available for organic fertilizers, so emission factors are estimated to be equivalent to those of mineral fertilizer. 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).

4.1.3.3 Plant availability of nutrients

The plant availability of the nutrients nitrogen, phosphorus, and potassium is a decisive factor to determine the effective potential of organic fertilizers from source-separation systems to substitute industrially produced fertilizer. Mineral fertilizers are usually developed to supply the total amount of nutrients in a short term. Thus, they can be applied precisely at a particular time to meet the nutritional demands of the agricultural crops, which is usually high during the growing period. The short-term availability is of particular importance for nitrogen and less important for phosphorus and potassium.

In general, multiple factors can influence the plant availability of nutrients from a certain fertilizer, including soil properties (pH, moisture content, oxygen, organic matter), time of application, and weather conditions (temperature and precipitation) (Finck, 1992). For this study, it is important to which extent organic fertilizers can effectively substitute industrially produced mineral fertilizers. The relevant figure is consequently a relative plant availability compared to mineral fertilizer:

$$\text{plant availability [\%]} = \frac{\text{plant availability of nutrient from secondary fertilizer}}{\text{plant availability of nutrient in mineral fertilizer}}$$

The relative plant availability of nutrients in mineral and organic fertilizers is different for each nutrient, because the biogeochemical cycles of the nutrients have distinct characteristics. A short background of the most important aspects of the fate of nitrogen, phosphorus and potassium in soil is given below.

Nitrogen

Nitrogen can be present either in a soluble inorganic form (NH_4-N and NO_3-N) or in organic material such as amino acids, proteins etc (N_{org}). Plants can only take up inorganic nitrogen, so the organic nitrogen has to be transformed in a soluble form ("mineralised") to be plant-available. The organic part of the nitrogen is therefore not completely accounted for when determining the nutrient equivalents of a certain fertilizer, because the mineralization of nitrogen can require a considerable time. Several biological processes can transform nitrogen in the soil, e.g. nitrification, denitrification etc. Pools of nitrogen in the soil include organic matter and ammonia absorbed in clay particles. Due to the complexity of the various interrelationships between physical, chemical and biological characteristics of soil, estimations of the plant availability of nitrogen highly depend on the specific conditions during fertilizer application (Bengtsson et al., 1997). Empirical field tests are a simple measure to estimate the relative plant availability of nitrogen from secondary fertilizers by comparing the yields for different types of fertilizer. Volatilisation of nitrogen species

has to be taken into account while determining the substitution potential of organic fertilizers to replace mineral fertilizers.

Phosphorus and potassium

For phosphorus and potassium, the short-term availability is less important for the fertilizing effect due to the more balanced demand of most crops throughout the year. Phosphorus is found in many inorganic forms as well as in constituents of soil organic matter and in living organisms. Only a very small part (less than 0.1% in soil solution) is present in the plant-available soluble forms of HPO_4^- and $H_2PO_4^{2-}$ (Bengtsson et al., 1997). Due to the relatively large pool of phosphorus in the soil, the availability of phosphorus for plant uptake is both a matter of chemical equilibriums and kinetics. Soil pH is one of the most important factors affecting the transformation of phosphorus in soil since these chemical processes are sensitive to changes in pH. Plant-availability of phosphorus may be limited in the short term if phosphorus is chemically fixed. Organic phosphorus has to be transformed into inorganic phosphorus by microbial activity before plant uptake. Due to the strong adsorption of phosphorus to soil particles, leaching is negligible for inorganic phosphate.

The plant availability of potassium in soil is determined by similar factors. A large pool of potassium is present in the soil in inorganic minerals (feldspar, clay etc) with a major fraction of irreversibly bound potassium. The kinetics of dissolution of potassium into the soil solution determines its plant availability. Potassium in organic fertilizers is assumed to be in a soluble form which is readily plant-available (Stadtmüller, 2004). Leaching of potassium from secondary fertilizers is comparable to that of mineral fertilizer.

Losses by volatilisation of phosphorus or potassium are neglected due to the physical characteristics of these inorganic ions and their species.

Plant availability

The relative availability of nutrients from secondary fertilizers has to be estimated according to pilot studies and field trials (Table 48). Values for plant availability of nitrogen are assumed after possible losses due to volatilisation of NH_3, N_2O, and NO_x. If no data is available for the specific substrate, figures for organic fertilizers with similar composition are applied.

Table 48: Plant availability of nutrients from organic fertilizers with regard to the substitution potential for mineral fertilizer

Substrate	Nitrogen	Phosphorus	Potassium	Source
	[% of total N*]	[% of total P]	[% of total K]	
Sewage sludge	10-63 / 50	20-70 / 70	100 / –	1 / 2
	50	**70**	**100**	
Urine	90-100 / 60-90 / 80-90	– / 80-120 / –	– / – / –	3 / 2 / 4
	100	**100**	**100**	
Biowaste compost	50 / 10 / 5-15	100 / 100 / 20-40	100 / 100 / >85	5 / 6 / 7
	10	**70**	**100**	
Digester residual (stabilised)	30 (20-60)			8
	30	**100**	**100**	
Digester residual (liquid)	*contains 100% of urine with high nutrient availability*			9
	90	**100**	**100**	

Bold: this study
**plant availability of nitrogen after volatilisation of NH_3, N_2O, NO_x, leaching of NO_3 to groundwater is not taken into account*
1) Schneidmadl, 1999 (literature review)
2) Bengtsson et al., 1997 (including volatilisation of nitrogen during storage and handling)
3) Peter-Fröhlich et al., 2007 (pot and field tests)
4) Stockholm Vatten, 2000 (field tests)
5) assumptions in Vogt et al., 2002
6) EPEA, 2004
7) Stadtmüller, 2004
8) Roschke, 2003 for dewatered digested manure
9) Jönsson et al., 2004 for blackwater

Sewage sludge

The plant availability of nutrients from sewage sludge depends on the chemical forms of the nutrients. If the sludge is stabilised and dewatered, a major part of the inorganic nitrogen is lost with the sludge liquor, leading to limited nitrogen availability. Typically, 25% of organic N and 90% of mineral N in sewage sludge are estimated to be plant-available (ATV, 1996). This study assumes a 50% availability of nitrogen for the dewatered sewage sludge. For phosphorus, the availability is heavily depending on the mode of P elimination in the wastewater treatment plant. While P in sludge from biological P elimination should be completely plant-available, chemically eliminated P may not be readily soluble due to the strong chemical fixation in the precipitates (Coker and Carlton-Smith, 1986; Bengtsson et al., 1997; Suntheim, 2001; IME, 2005).

Therefore, a limited P availability of 70% is assumed for phosphorus from chemical P elimination in this study. However, the overall plant availability can eventually reach 100% in the long term (ATV, 1996; Onnen, 2001; IME, 2005).

Urine

For urine, the availability of all nutrients seems to be equal to that of mineral fertilizer. Results from pot and field tests in Germany revealed no significant difference to industrial fertilizer in terms of nitrogen availability (Peter-Fröhlich et al., 2007; Simons and Clemens, 2004), presumably due to the fast transformation of urea into ammonium by microbial activity. Other studies assume a reduced availability of nitrogen in urine (60-90%), but here losses during storage and handling (volatilisation and leaching) are included in the figures. Phosphorus and potassium are present in their soluble inorganic form which is readily plant-available.

Biowaste compost

In biowaste compost, the major part of nitrogen is bound to the organic matrix due to high microbial activity. Inorganic nitrogen is mainly lost by volatilisation of NH_4-N to the atmosphere during the composting process and NO_3-N in leachate, leading to low nitrogen efficiency (10%). The P availability is estimated to be limited to 70% due to the high fraction of organic phosphorus (Stadtmüller, 2004).

Digester residual

For the residual of the co-digestion process of faeces and biowaste, the nitrogen efficiency particularly depends on the post-treatment. If the digester residual is dewatered and further stabilised by open composting/storage, a large part of the plant-available inorganic nitrogen is lost with the leachate (NO_3-N) or through atmospheric emissions (NH_4-N) similar to biowaste compost. Literature values for dewatered digester residual from animal manure suggest that only 30% of the remaining nitrogen can be used by the plants (Roschke, 2003). Phosphorus and potassium is also lost to a smaller extent with the leachate, but the phosphorus remaining in the composted residual should be completely usable for the crops.

In case of direct application of the liquid digester residual, the nitrogen availability is estimated to be almost comparable to mineral fertilizer (90%). Data for the fertilizing equivalence of digested blackwater is scarce. Pilot studies indicate a reduced N availability for undigested blackwater (Simons and Clemens, 2004). A major part of the nutrients contained in this residual derives from the urine, for which a complete nutrient efficiency is assumed. The digestion process itself increases the plant-available nitrogen by the conversion of N_{org} into NH_4-N without significant losses of gaseous nitrogen. Consequently, liquid digester residual is assumed to be a good fertilizer with high nutrient availability.

4.2 System construction

The implementation of a source-separation sanitation system has not been realized to date in an urban settlement with 5000 inhabitants. Pilot plants for several houses or small settlements (10 – 150 inhabitants) have been built in Sweden (Johansson et al., 2001) and Germany (Oldenburg et al., 2002; Otterwasser, 2005; Peter-Fröhlich et al., 2007), but related construction data – if available – cannot be directly transferred to a larger settlement.

Thus, construction data has to be generated for the source-separation systems. For reasons of symmetry, construction data of the conventional system is generated with the same methodology, even though respective data for the construction of conventional sewers and treatment would be available. The housing structure of the settlement is adopted from an existing quarter of the Berlin area (Berlin-Nikolassee) to reflect realistic conditions. Data from the geographical information system (GIS) of the local water supplier is used for the layout of the sanitation system. Conventional and source-separation systems have been exemplary designed in close cooperation with consulting engineers (Otterwasser GmbH, Lübeck, Germany) and have been published before (Peter-Fröhlich et al., 2007). Consultants are experts with considerable experience in the design of source-separation systems (e.g. Lübeck-Flintenbreite).

4.2.1 Settlement structure

The settlement structure is adopted from an existing quarter of Berlin with approxiamtely 5000 inhabitants. The quarter is located in the south-west of the Berlin city area and can be characterized as a sub-urban area (Berlin-Nikolassee). The distribution of houses, house types and other structural data are taken from data of the geographical information system (GIS) of Berliner Wasserbetriebe (Berlin water works, Table 49). The quarter is divided by a large road into a northern part and a southern part (see annex 12.6.1 for a map).

The layout of the sewer systems and the treatment facilities is based on the real distribution of inhabitants throughout the area. For simplification of sewer dimensioning and slope, it was assumed that the whole terrain is plain (i.e. no elevation or hills).

Table 49: Structural data of the urban settlement

Parameter			Remarks
Population	5000	*inhabitants*	
Area	126	*ha*	
Population density	40	*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*	= 1170 apartments (assumption: three-storey buildings with three apartments per floor)

Data adopted from GIS data of existing city quarter (Berlin-Nikolassee)

4.2.2 Inventory

The construction inventory of the sanitation systems includes sanitary in-house installations, the complete sewer system, and the treatment facilities. Although the wastewater system layout can heavily depend on local conditions, the following calculations are supposed to provide a defensible estimation of the different infrastructural needs of the scenarios. More details of the construction inventory are provided in annex 12.6.

4.2.2.1 Sanitary in-house installations

The realization of the sanitary in-house piping strongly depends on the particular layout of the respective buildings. For reasons of simplicity, a prototype layout for a house unit and an apartment unit is assumed by estimating necessary pipe lengths and diameters. The dimensioning is done in accordance with the relevant legal regulations in Germany (DIN EN 12056-2, 2000).

Pipe lengths and diameters are multiplied with the number of houses and apartments to calculate the total material demand (Table 50). In-house piping is made from polypropylene (PP) or high-density polyethylene (HD-PE) for houses and cast iron for apartments (due to legally required fire prevention measures). The additional weight of fittings and plug-in connections is estimated with a proportional factor. The sanitary in-house piping includes the base pipe below the basement which runs up to the house shaft.

Table 50: Inventory data for pipes of sanitary in-house installations

Material	Ø	Weight*	*WW*	*GW*	*SWS*	*F*	*Vac***	*U*	Fitting factor***
	[mm]	*[kg/m]*	*[m]*	*[m]*	*[m]*	*[m]*	*[m]*	*[m]*	*[%]*
PP	50	0.285	12180	13050				2910	*15*
	70	0.454		6253				8948	*15*
	100	0.938	10875	7246		10875		6300	*15*
HD-PE	20	0.172			18156				*10*
	50	0.666					17070		*10*
	150	1.943	6300			6300			*10*
Cast iron	50	4.3	7020	7020					*10*
	70	5.9		3510					*10*
	100	8.4	6338			6338			*10*
Total length		*[m/pe]*	8.5	7.4	3.6	4.7	3.4	3.6	
Total plastics		*[kg/pe]*	5.8	3.1	0.7	5.0	3.76	2.5	
Total cast iron		*[kg/pe]*	18.4	11.2		11.7	0.6		

WW: combined wastewater, GW: greywater, SWS: service water supply (for greywater reuse), F: gravity drainage of faeces, Vac: vacuum system, U: urine
** Sources: Ostendorf, 2005; Simona, 2005; Dueker, 2005*
*** plus vacuum collection tanks: 1260 units (each with 5 kg PE and 2.2 kg cast iron)*
**** additional factor to account for fittings (10%, for PP: + 5% for plug-in connections)*

4.2.2.2 Sewer system

The sewer system includes house shafts, house connections to the main sewer, and the sewer system with piping and inspection chambers, ending at the respective treatment facilities. The sewer system design follows common German rules in sewer layout (ATV, 1999; DIN EN 752, 1997). The pipes of the conventional system have the largest diameter (DN 150 - 400) due to the high volume of combined wastewater. For the source-separation scenarios, the dimensioning of the gravity drainage systems is adjusted to the reduced volume of greywater or toilet wastewater. The urine collection system has a separate pipe for gravity drainage of separated urine to pumping shafts, from where it is pumped to holding tanks. Purified greywater is supplied via service water net for non-potable reuse.

The resulting total length of the sewer pipes for each sanitation system is presented in Table 51. Materials for the pipes are vitrified clay (combined wastewater, toilet wastewater) or plastics (greywater, service water, urine, vacuum, pressure pipes).

Table 51: Total length of sewer systems

System	Material	House connections	Drainage		Pressure pipes**	
			Diameter	*Length*	*Diameter*	*Length*
		[m DN 150]	*[mm DN]*	*[m]*	*[mm DN]*	*[m]*
Combined wastewater	Vitr. clay	10000	150-400	13805		
Greywater	PE	10000	150-200	13638		
Service water supply	PE	10000*			150	11108
Faeces	Vitr. clay	10000	150	11108	50	7816
Vacuum	PE	10000*	65-100	13308	65	2241
Urine	PP	10000	150	11108	50	7050

** DN 32 for service water supply, DN 50 for vacuum system*
*** Faeces: to solid-liquid separation, vacuum: to biogas plant, urine: to holding tanks*

Major additional parts (i.e. inspection chambers, pumping shafts and pressure pipes) are included in the construction inventory. Material demand for pipes and additional parts is aggregated to calculate the total material demand for the sewer system (Table 52). The amount of soil material from excavation is calculated from the required depth of the piping, providing a sufficient slope for gravity drainage. Pipes of source-separation systems are laid in one combined trench wherever possible to minimize the required excavation volume.

Table 52: Total length and total material demand for sewer systems per person

System		WW	GW	SWS	F	Vac	U
Total length	*[m/pe]*	4.8	4.7	4.2	5.8	5.1	5.6
Total material demand*							
Vitrified clay	*[kg/pe]*	145	0	0	111.5	0	0
PP/PE	*[kg/pe]*	1.3	13.6	12.6	1.9	4.1	12.5
Concrete	*[kg/pe]*	552.4	508.6**	0	20	18.8	20
Cast iron	*[kg/pe]*	25.6	21.7**	0	1.2	1.7	1.2
Excavation	*[m³/pe]*	8.9	9**	0	0.1	0.1	0.4

WW: combined wastewater, GW: greywater, SWS: service water supply (for greywater reuse), F: gravity drainage of faeces, Vac: vacuum system, U: urine
** including pipes + fitting factors, sealing, inspection chambers, and pumping shafts*
*** house shafts, inspection chambers and excavation for source-separation systems are assigned to greywater system*

The comparison of the calculated total length of the sewer system and the in-house installations for each scenario reflects the multiple piping that is required in source-separating systems (Figure 32). Two-flow systems (scenarios V) and three-flow systems (i.e. with urine separation, scenarios SC and SV) enhance the necessary sewer length by a factor of 2 or 3, respectively. Greywater reuse requires another piping system for the recycling of purified greywater to the households. For the in-house installations, the additional piping is less than proportional to the number of separated flows due to constructional reasons (e.g. one ventilation pipe for multiple systems).

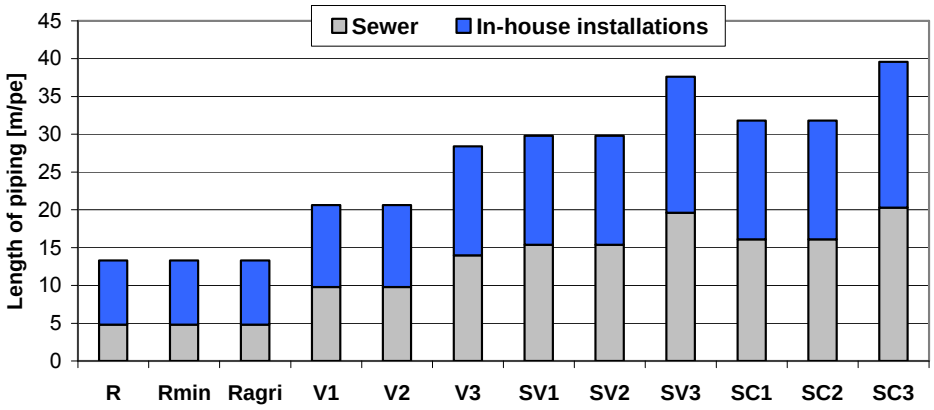

Figure 32: Total length of sewer systems and in-house installations per person for each scenario

4.2.2.3 Treatment facilities

The construction expenditures for the treatment facilities in the different sanitation scenarios are estimated by a simplified layout. Major parts of the systems are included in the inventory (Table 53), whereas smaller parts are neglected. Only the composting plant is excluded from the construction inventory due to a lack of data. However, composting plants are usually large facilities for high volume throughput, so that most likely existing plants would be used for biowaste and faeces treatment.

Adequate inventory data for construction is not directly available for most of the facilities, so inventory data has to be adopted from other LCA studies or pilot plants. Where no comparable data could be acquired, material data is roughly estimated.

Table 53: Treatment facilities included in the construction inventory

System	WW	GW	SWS	F	Vac	U
	Conventional activated sludge plant with digester	SBR or MBR or soil filter	Storage tanks	Solid-liquid separation	Vacuum system Biogas plant	Urine holding tanks
	Service buildings for source-separation systems					

WW: combined wastewater, GW: greywater, SWS: service water supply (for greywater reuse), F: gravity drainage of faeces, Vac: vacuum system, U: urine

Inventory data for the particular facilities is compiled from the following sources:

- *Conventional activated sludge plant:* A detailed inventory is adopted from an LCA study on sanitation systems (Schneidmadl, 1999). Inventory data originates from an activated sludge plant with digester (21000 inhabitant equivalents) and is recalculated in proportion to the influent volume.
- *SBR for greywater treatment:* plant data is recalculated from data of the conventional activated sludge plant with reduced daily influent volume
- *MBR for greywater treatment:* plant data is basically adopted from activated sludge plant. The compact construction of this process is taken into account by reducing the material demand for tank construction (concrete, non-alloy steel, excavation) by a proportional factor. Typically, aeration tanks for the activated sludge process can be smaller by a factor of 3 – 4 for an MBR process, and the process does not need a final clarifier (Pinnekamp and Friedrich, 2006). Additional material demand for membranes and other equipment is neglected.

- *Soil filter:* Inventory data is adopted from material specifications of the pilot plant in Lübeck-Flintenbreite (Oldenburg, 2002) and related to the required surface area. Sedimentation tanks are additionally provided as primary treatment (HRT > 3 h).
- *Greywater reuse:* Storage tanks for purified greywater prior to reuse (= service water) are designed to provide the average service water demand of one day (Fbr, 2005).
- *Urine tanks:* Holding tanks for urine have a sufficient volume to store separated urine of 14 days as a minimum (= 21 L/pe).
- *Vacuum system:* Inventory is estimated from manufacturer information for steel tanks and pumps.
- *Biogas plant:* Inventory data for the biogas plant is adopted from other LCA studies (Edelmann et al., 2001; Ronchetti et al., 2002) and recalculated in proportion to the required digester volume.
- *Solid-liquid separation:* The equipment for the solid-liquid separation process for faeces dewatering is not yet commercially available, so the required material is roughly estimated.
- *Service buildings:* For source-separation systems, three service buildings (70 m²) are provided in the settlement to contain the various treatment facilities (vacuum plant, solid-liquid separation etc).

In general, the construction inventory for the treatment facilities heavily depends on actual design and local boundary conditions. The construction data calculated in this study can only provide a rough estimation of the material demand for facility construction. However, the facilities constitute only a minor part compared to in-house piping and sewer system, so the uncertainty in this part of the inventory should not compromise the overall results significantly.

4.2.3 Service life

In order to allow the aggregation of the inventory data from construction and operation of the sanitation systems, the construction expenditures 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 sanitation 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.

In this study, service life for the different components of the sanitation systems is estimated from other LCA studies in this field (Bengtsson et al., 1997; Schneidmadl, 1999) and data for economic purposes (LAWA, 2005). The chosen values reflect a more conservative approach and are probably smaller than the realistic material lifetimes (Table 54). On the other hand, this LCA study considers the construction of sanitation systems, but excludes expenditures for both maintenance and repair. These exclusions are neglecting the effort necessary to reach a long service life for the sanitation systems. The assumption of a relatively short service life may roughly offset these limitations. While evaluating the influence of the construction phase on the results of the LCA, the question of adequate assumptions for service life has to be kept in mind.

Table 54: Estimated service life of infrastructure components

	Bengtsson et al., 1997	Schneidmadl, 1999	LAWA, 2005	This study
Drainage		*Service life in years*		
In-house pipes	30	–	–	**40***
Sewer pipes	30	75	50 – 80	**50***
Manholes/shafts	–	–	50 – 80	**50***
Pumps	15	15	8	**12.5**
Facilities				
Buildings	30	–	30 – 50	**40**
Tanks	30	35	30 – 40	**40**
Machinery	15	15	8 – 30	**12.5**
Vacuum plant	15	15	25 – 40	**40**
Soil filter	30	50	12 – 15	**40**

** independent of material*

4.2.4 Materials for system construction

The components of a sanitation system are made from different materials. Sanitary installations inside the houses are mainly made of cast iron or plastic, whereas drainage pipes of the sewer system are usually 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.

Most of the inventories for material supply are delivered with Umberto® software (IFU and IFEU, 2004). They are based on relatively old LCI datasets and are not described in detail in this report. Other LCI data sets (e.g. steel, copper) are developed from literature data of material and energy flows by connecting them to background datasets from Umberto®. More recent LCI datasets (e.g. from ecoinvent database) were not accessible by the author of this study. The transport distance for all materials from the supplier to the settlement is assumed to be 300 km. For concrete, a shorter transport distance of 50 km is assumed which is done in accordance to other LCA studies (e.g. Reckerzügl, 1997; Frischknecht and Jungbluth, 2002). Concrete production is typically located close to a building site to minimize transport costs for this heavy material which is usually required in high amounts.

Aside from the production of building materials, the excavation of trenches forms another major part of the construction expenditures. The energy demand for excavation is calculated assuming a demand of 115 g diesel fuel for 1 m^3 of excavated material (Frischknecht et al., 1996), which is equivalent to an amount of mechanical energy of 1.425 MJ per m^3 soil (IFU and IFEU, 2005). The operational emissions of the excavator are calculated with a dataset for the operation of construction machinery (IFU and IFEU, 2005).

Table 55: Construction materials and inventory datasets

Material	Components	Source	Remarks
Pipes			
Polypropylene	In-house pipes, urine pipes	1	Boustead, 1999a
Polyethylene	Sewer pipes, vacuum pipes	1	Boustead, 1999a + welding energy
Cast iron	In-house pipes	1,3	Recycling share: 88%
Vitrified clay	Sewer pipes	1,3	Including sealing
Other			
Concrete	Manholes, tanks, buildings, digester	2	
Construction steel	Buildings, tanks	2,4	Recycling share: 42%
Stainless steel	Machinery, vacuum plant	2,4	Recycling share: 42%
Cast iron	Manhole covers, pumps	1,3	Recycling share: 88%
Copper	Activated sludge plant	2,5	
Aluminium	Activated sludge plant	1	Boustead, 2000
Polypropylene	Urine pumping shafts	1	Mouldings
Polyethylene	Activated sludge plant, soil filter	1	Mouldings
Glass fibre reinforced plastic	Urine tanks, service water storage	1	Epoxy resin and glass (1:1)
Limestone	Activated sludge plant	1	Patyk and Reinhardt, 1997
Excavation	Trenches, tanks	1	1.425 MJ E_{mech}/m³ soil, incl. excavator emissions

Transport to construction site is 300 km by truck for all material except concrete (50 km)
Sources:
1) IFU and IFEU, 2005
2) Frischknecht et al., 1996
3) Jeschar et al., 1995; Jeschar et al., 1996
4) BDSV, 1998; Corradini and Köhler, 1999; Fritsche et al., 2001
5) Kraus et al., 1999; Kippenberger, 2001

4.3 Background processes

Background processes are processes or services that are included within the system boundaries, but do not belong to the core system under investigation. Here, background processes include the supply of electric and thermal energy, transport by truck, waste incineration, and the supply of auxiliary materials for system operation (e.g. chemicals). Additionally, the system expansion process of industrial fertilizer production is described in this section.

4.3.1 Energy Supply

The supply of energy includes the forms of electric and thermal (heat) energy. The datasets are related to conditions in Germany in terms of power plant technology and energy mix.

Electric energy
The expenditures for the production and supply of electric energy are calculated with datasets provided by Umberto® (IFU and IFEU, 2005). These datasets are mainly based on data from databases GEMIS (Fritsche et al., 2001) and ECOINVENT (Frischknecht et al., 1996). 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.

The technology of the different types of power plants represents German conditions of 1990 – 2000. Data for the German energy mix in 2003 (Table 56) is derived from the federal ministry of economics (BMWi, 2005). Heating values for the different fuels are assumed according to Umberto® datasets (Table 56). It is further 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 total energy production (Frischknecht et al., 1996).

Table 56: Power mix and 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 (1.7%)
2) electric energy output to thermal input (heating value)
3) density 0.776 kg/m³

Thermal energy

The supply of thermal energy is described with Umberto® datasets based on GEMIS data (Fritsche et al., 2001). Power plant technology represents German conditions from 1990. 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 electric energy production, as most power plants produce both types of energy. For this study, thermal energy is produced in dedicated heating plants, and an average fuel mix is assumed (Table 57).

Table 57: 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

** thermal output to thermal input (heating value)*

4.3.2 Transport by truck

The transport processes in this study are described with the respective dataset for truck transport from Umberto® (IFU and IFEU, 2005). This dataset is based on the TREMOD model which was developed for the German environmental agency (Knörr et al., 1997). The parameterized model calculates fuel consumption and emissions as a function of the transport distance, the cargo weight, the load factor, the type of truck, and the road categories. The dataset is based on truck stocks operated in Germany in 1996.

In this study, only the transport distance and the cargo weight are adjusted depending on the respective transport process in the system. The remaining parameters of the dataset are assumed to be constant for each transport process. The load factor is set to 100% (= fully loaded) for one way and 0% (= empty trip) for the return. The truck type is assumed to be a single truck with a total weight > 20 tons. The distribution of road categories is set to 37% on highways, 41% on country roads, and 22% on municipal roads according to average German data for this type of truck (IFU and IFEU, 2005). With these assumptions, the TREMOD dataset calculates an overall primary energy demand of 2 MJ per ton*km for the return trip, including the pre-chain for the provision of fuel.

The transport distances of the respective cargo are heavily depending on local boundary conditions. In an urban context, the distance of suitable farmland for application of organic fertilizers can be relatively high, whereas it should be shorter for a rural setting. If animal manure is readily available or has to be disposed, the demand for other organic fertilizers in the proximity can be limited, resulting in longer transports distances. Especially the transport of high-volume organic fertilizers (e.g. urine) may eventually have a high influence on the energy balance of the system. Consequently, this parameter is varied in sensitivity analysis to quantify its impact on the results of the study.

The preset distances for all transport processes are estimated according to comparable LCA studies and own assumptions (Table 58, cf. annex 12.7):

- The transport of industrial products (mineral fertilizer, precipitation chemicals, and construction materials) is assumed with 300 km, except for concrete which is typically produced closer to the construction site (50 km)
- Composting plants and waste incineration plants are large facilities with wide service areas, resulting in a longer transport distance (20 and 30 km, respectively)
- Biogas plants are smaller facilities that can be located close to the settlement, so that truck transport is not necessary at all if pipes are used for the substrates
- For the organic fertilizers, a distance of 20 km is assumed from the treatment facilities (WWTP, composting or biogas plant, urine treatment) to the farms

- The urine collection and transport to the ozone treatment is estimated to 5 km
- Biowaste from households is collected in a stop-and-go mode before it is transported to the composting plant. Umberto® provides a dataset for biowaste collection with an average distance of 7.5 km in stop-and-go mode

Table 58: Transport processes and distances

Cargo	From	To	Distance
Fertilizers			*[km]*
Mineral fertilizer	Manufacturer	Farms	300
Sewage sludge	WWTP	Farms	20
Biowaste and faeces	Collection*	Compost plant	20
Biowaste, faeces	Collection*	Biogas plant	0
Compost	Compost plant	Farms	20
Digester residual	Biogas plant	Farms	20
Urine	Holding tanks	Ozone treatment	5
	Ozone treatment	Farms	20
Waste			
Sewage sludge	WWTP	Incineration	30
Biowaste	Collection*	Incineration	30
Auxiliary			
Chemicals	Manufacturer	WWTP	300
Construction materials			
Concrete	Manufacturer	Settlement	50
	Settlement	Disposal	50
Plastics, metals, etc	Manufacturer	Settlement	300
	Settlement	Disposal	100

** plus stop-and-go collection of biowaste (7.5 km)*
Sources: annex 12.7

4.3.3 Incineration plant

The co-incineration of biowaste or dewatered sewage sludge in an incineration plant for domestic waste is described by an Umberto® dataset for a municipal solid waste incineration plant (IFU and IFEU, 2005). It represents an average technology of the German plants in 1990 (grate firing, four-step flue gas cleaning with electrostatic filter, spray absorber, baghouse filter with coke as absorbent, and Denox catalytic converter). The dataset calculates the demand for auxiliary material and energy, the emissions and residues, and the electric or thermal energy derived from the incineration of the waste.

The calculation is based on the amount and elemental composition of the waste and its heating value.
The heating value of the biowaste or sewage sludge is calculated roughly via Dulong´s formula from the elemental composition (including C, H, O, S, H_2O). 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 (light heating oil) 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 supply of auxiliary material (coke, lime, light fuel oil) required for plant operation is included with datasets from Umberto® (IFU and IFEU, 2005).

4.3.4 Auxiliary material

Data for the supply of auxiliary materials is based on Umberto® datasets (IFU and IFEU, 2005. Flocculants ($FeCl_3$, $FeSO_4$) are produced from industrial waste acids with energy demand as the most relevant input (Ruhland, 2004). Coagulation aid (polyacrylamide) is produced via polymerisation of acryl amide which is synthesized from acrylonitrile. Coke, lime, and light fuel oil are required for the operation and flue gas cleaning of the incineration plant. The respective datasets are calculated with Umberto® data for basic materials, connected to specific literature data of production processes (Table 59). The relevance of auxiliary materials for the present study is limited, so that details of these datasets are not provided here. Further information can be found in other LCA studies (Ruhland, 2004).

Table 59: Auxiliary materials for system operation

Material	Application	Source	Remarks
Ferric chloride (40% solution)	Flocculant	1)	Production from waste acid, scrap iron and chlorine (CED = 2.74 MJ/kg $FeCl_3$)
Ferrous sulfate (96% solution)	Flocculant	1)	Drying of $FeSO_4$ (214 kWh/t $FeSO_4$)
Polyacrylamide	Coagulation aid	1)	Based on data for acrylonitrile production (Boustead, 1999b)
Lime	flue gas cleaning in waste incineration plant	1)	Calcination of limestone + hydration
Coke		2)	Based on Corradini and Köhler, 1999
Light fuel oil	additional fuel in incineration plant	2)	Based on Fritsche et al., 2001 and Frischknecht et al., 1996

1) Ruhland, 2004
2) IFU and IFEU, 2004

4.3.5 Industrial fertilizer production

Numerous different products of single or multi-nutrient fertilizers are produced in the fertilizer industry, and they offer a range of different nutrient contents. The main macronutrients are nitrogen (quantified 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 can 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.

Heavy metal contents

For calculating the heavy metal content of an average P, N, or K fertilizer related to its nutrient content, the nutrient concentration and market shares of the different fertilizers have to be combined with their average heavy metal content.

Data for the average nutrient content of mineral fertilizers (Patyk and Reinhardt, 1997) is combined with market shares in the German fertilizer market in 1998/99 (Drescher-Hartung et al., 2001). The average heavy metal contents for each fertilizer are compiled from several studies (Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001), where the original source relates to the year 1992 (Boysen, 1992). Thus, calculated mean concentrations of heavy metals (Table 60) are based on relatively old data. A comprehensive up-to-date dataset for heavy metals in mineral fertilizers is not publicly available.

Table 60: Mean concentrations of heavy metals and As for average mineral fertilizers, related to the single nutrients

[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

Sources: Patyk and Reinhardt, 1997; Drescher-Hartung et al., 2001; Hackenberg and Wegener, 1999 (see annex 12.8.1 for calculation details)

Recently, new values for heavy metal contaminations of several mineral fertilizers sold in the EU were published (UBA, 2007a), suggesting significant changes for certain heavy metals compared to the dataset from Boysen (cf. annex 12.8.1). However, UBA data is not considered to be representative for the German fertilizer market due to the low number of analysed samples. Thus, it is decided to use comprehensive data from Boysen despite its relatively high age (15 years). The effect of using updated heavy metal data from UBA on the toxicity assessment is shown in sensitivity analysis.

The contamination of phosphate fertilizer with elevated levels of elemental Cd, Cr, and Zn originates mainly from raw phosphate ores. Recently, the toxic element uranium (U) was detected in significant concentrations in mineral phosphate fertilizers (Kratz, 2004; Fink, 2005). Due to lacking LCIA characterization factors for its toxic effects, uranium is not included in the impact assessment method of this study.

Inventory for production

The production of mineral fertilizers in Germany and the associated substance and energy flows have been carefully documented (Patyk and Reinhardt, 1997). However, emissions in surface waters are not listed in this data, although they may be important for the environmental evaluation. 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. Aquatic emissions of phosphate and fluoride are updated with recent data from other studies (see annex).

The inventory 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 61). The transport of mineral fertilizers from the production to the customer is not included.

Table 61: Life cycle inventories of mineral fertilizer production

Reference value		1000 kg N	1000 kg P_2O_5	1000 kg K_2O
Input				
Use of resources				
Raw potash	*kg*			10500
Limestone	*kg*	550		
Raw phosphate ore	*kg*		4060	
Sulphur	*kg*		272	
Cumulated energy demand	*MJ*	48896	17432	10382
Output				
Emissions (air)				
CO_2 (fossil)	*kg*	2820	1117	617
CO	*kg*	2.80	1.42	0.42
CH_4	*kg*	7.45	2.07	1.38
NMVOC	*kg*	0.54	0.49	0.12
NH_3	*kg*	6.69	0.01	0
N_2O	*kg*	15.05	0.04	0.05
NO_x	*kg*	15.76	8.58	1.15
SO_2	*kg*	5.16	11.98	0.27
Dust (> PM10)	*kg*	2.31	1.11	0.85
HF	*kg*	0	0.023	0
Emissions (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
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	4.18	0.00
Pb	*g*	2.67	19.58	0.19
Zn	*g*	4.95	27.48	0,27
Nutrients				
NH_3	*g*	2.68	9.17	1.72
NO_3	*g*	189.15	8.16	1.20
PO_4	*g*	28.62	*4400	1.40
Chloride	*g*	6219	5827	825.00
Fluoride	*g*	1.65	*2200	0.27

Sources: Patyk and Reinhardt, 1997; Gaillard et al., 1997 for aquatic emissions
** updated with other data, see annex 12.8.2 for details*

5 Results

This chapter presents the results of the Life Cycle Assessment along with a discussion of relevant findings. This includes:

- selected results on the level of Life Cycle Inventory
- all indicator results from the Life Cycle Impact Assessment with contribution analysis
- normalisation of indicator results to total impacts in Germany to reveal their relative contribution to the respective impact categories
- exemplary procedure of grouping and weighting of indicators to come up with a conclusive evaluation while comparing two scenarios
- sensitivity analysis to test the stability of the results against the variation of important system parameters

5.1 Selected results of the Life Cycle Inventory

The setup of the Life Cycle Inventory (cf chapter 4) results in the collection of a huge amount of data for all relevant processes. Input-Output balances can be compiled for each single process or for the entire system under investigation. Due to the large number of sub-processes and scenarios, these inventory results cannot be shown entirely in this study. However, it can be useful to show selected results of the Life Cycle Inventory on the level of cumulative emissions or resource demand before these results are transferred into indicators during Life Cycle Impact Assessment. On the one hand, inventory results can give a first hint of important differences between the investigated systems. On the other hand, the consistency of the inventory data can be checked, and the indicator results of the impact assessment are more transparent and comprehensible. Detailed data on the level of LCI is listed in annex 12.9.

Selected results of the Life Cycle inventory are presented here for:

- demand of electric energy for system operation
- supply of nutrients to agriculture with organic fertilizers
- system expansion processes (industrial fertilizer and grid energy)
- effluent loads of COD, nitrogen and phosphorus from wastewater and greywater treatment plants
- heavy metal loads emitted to surface waters and agricultural soil

5.1.1 Demand of electric energy for operation

Many processes require electric energy during the operation of the different sanitation systems. The input-output balance for each scenario shows the composition of electric energy input (= demand) and output (= supply), separated for the different sub-processes (Figure 33). The resulting net energy demand gives a picture of the effective demand for electric energy in each scenario (Figure 34).

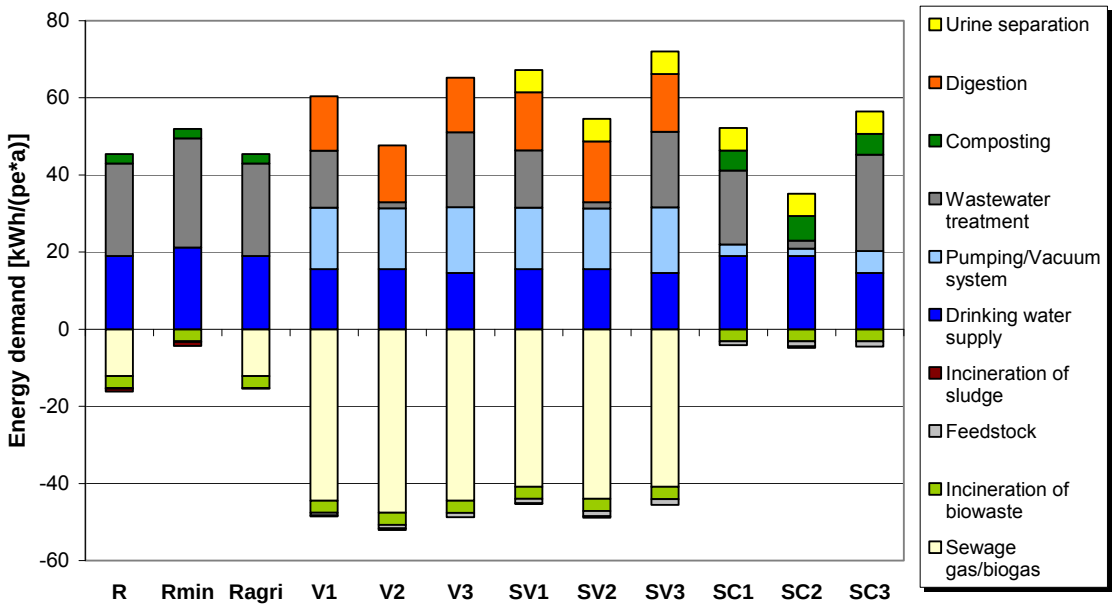

Figure 33: Input-Output balance of electric energy for system operation

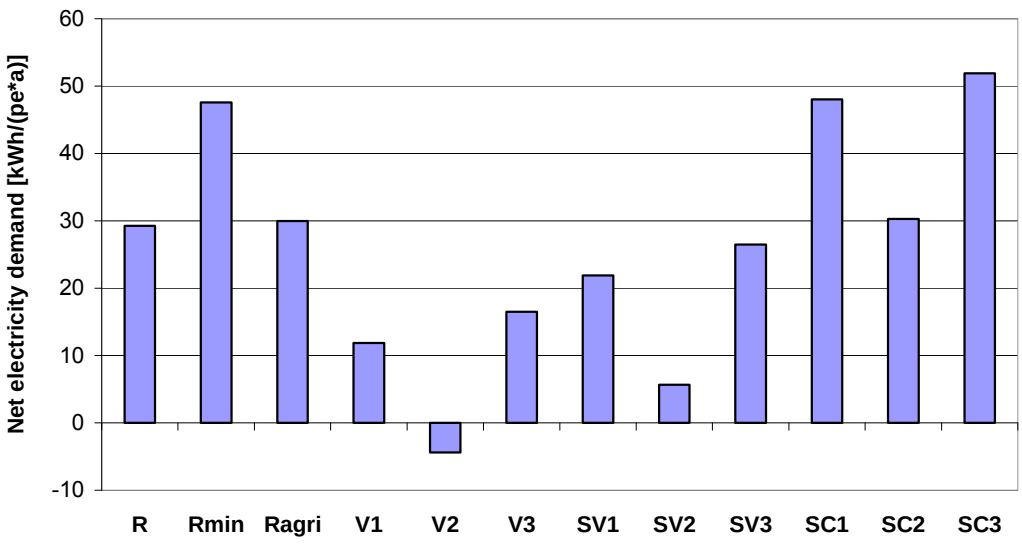

Figure 34: Net electricity demand for system operation

Reference scenarios

In the reference systems, a major part of the demand is due to the supply of drinking water (~ 40%) and the treatment of combined wastewater (~ 55%). The use of low-flush toilets lowers the demand for drinking water demand by 12% (= 2 kWh/(pe*a)) in scenarios R and R_{agri} compared to the R_{min} scenario. The energetically optimized wastewater treatment in scenarios R and R_{agri} leads to energy savings of 16 kWh/(pe*a), of which 4 kWh/(pe*a) are due to more efficient machinery and 12 kWh/(pe*a) are due to energy recovery via sludge digestion with sewage gas electrification. Thus, the net energy demand of the R_{min} scenario can be lowered from 48 kWh/(pe*a) to 30 kWh/(pe*a) (-38%) if the wastewater treatment is energetically optimized and low-flush toilets are used.

Separation scenarios

For the separation systems, the input of electric energy is generally higher than for the reference system. Even though the use of low-flush or vacuum toilets saves some energy in drinking water production (~ 3 – 7 kWh/(pe*a)), additional processes such as the vacuum system (15 kWh/(pe*a)), the operation of the digestion process (14 – 17 kWh/(pe*a)) or urine treatment (6 kWh/(pe*a)) offset these savings and increase the total energy input compared to the conventional wastewater treatment system (Figure 33).

Energy recovery

However, the potential for energy recovery via digestion of faeces and biowaste is substantial. In digestion scenarios, 41 – 48 kWh/(pe*a) can be recovered via biogas production (= 63 – 109% of the total demand for electric energy in scenarios SV and V). It has to be noted though that a substantial part of the recovered energy does not originate from wastewater constituents itself, but from the organic biowaste. In fact, 23 kWh/(pe*a) of the energy from the biogas plant is due to the digestion of biowaste, corresponding to 49-57% of the total recovered energy. Offsetting this energy with the energy demand for biowaste pretreatment for digestion (6 kWh/(pe*a)), a net energy potential of 17 kWh/(pe*a) can be allocated to the biowaste. This part of the energy recovery potential could also be used in conventional wastewater treatment systems if biowaste would be treated in an external biogas plant.

Scenarios with faeces composting (SC) do not use the energy potential of the organic matter. Consequentially, composting scenarios are comparable (SC2) or higher (SC1 and SC3) in net energy demand in relation to the reference system, while digestion scenarios offer the potential for significant energy savings.

Greywater treatment
The technology for greywater treatment is another decisive issue for the net energy demand of separation systems. Obviously, natural treatment of greywater in soil filters has by far the lowest energy demand (~ 2 kWh/(pe*a)) followed by greywater treatment in SBR (15-19 kWh/(pe*a)) or MBR plants (19-25 kWh/(pe*a)). In comparison, conventional wastewater treatment requires 24-28 kWh/(pe*a) in the reference scenarios. Hence, separation scenarios with soil filter and energy recovery (V2, SV2) are those systems with the smallest net demand of electric energy.

Greywater reuse
The non-potable reuse of treated greywater for the substitution of drinking water does not lead to an energetic benefit for the reuse scenarios V3/SV3/SC3. The substituted amounts of toilet flush water are too small (5-24 L/(pe*d)) to equalize the higher energy demand of the MBR process. Additionally, pump energy for delivering purified greywater back to the households is estimated rather high (= 0.23 kWh/m³), corresponding to about 50% of the energy demand for drinking water supply (= 0.5 kWh/m³). Hence, the reuse scenarios are not energetically favourable under these conditions. It is decided to further investigate this issue in sensitivity analysis to reveal the conditions (e.g. amount of substituted drinking water, energy demand for drinking water supply) in which the non-potable reuse of greywater becomes energetically beneficial.

In all, the separation scenarios with energy recovery (digestion of faeces and biowaste) offer potentials for a substantial decrease in the net energy demand of wastewater treatment. If energy from organic matter of faeces and biowaste is fully recovered, the net energy demand can be reduced by 30-60% with greywater treatment in an activated sludge process, resulting in a net energy demand of 12-21 kWh/(pe*a). If a low-energy process is used for greywater treatment (soil filter), the net energy demand amounts to only ~ 6 kWh/(pe*a). The lowest energy demand of all scenarios is equivalent to a net energy surplus (+ 4 kWh/(pe*a)) in the scenario V2 with energy recovery and low-energy greywater treatment. In this scenario, the supply of drinking water and the treatment of wastewater can even be achieved with a small overall energy benefit.

In general, the higher energy demand for the operation of separation systems is offset by the energy recovered from the organic matter of faeces and biowaste. About half of the energy recovery potential originates from biowaste. Separation scenarios without energy recovery (composting of faeces and biowaste, SC scenarios) are not energetically favourable to the conventional system. In fact, these systems tend to require more energy than the optimized conventional system. Similarly, the non-potable reuse of greywater does not lead to energetic benefits under the boundary conditions specified in this study.

5.1.2 Supply of organic fertilizers

The amount of plant-available nutrients nitrogen, phosphorus and potassium supplied in the form of organic fertilizers varies significantly between the scenarios (Figure 35). It depends on the extent in which wastewater nutrients are contained in or transferred to the respective fertilizer, possible losses during treatment, handling and application, and the physical and chemical properties of the organic fertilizer (e.g. availability of nutrients for plant uptake).

Reference scenarios

In the reference scenarios without agricultural application of sewage sludge (R and R_{min}), wastewater-derived nutrients are completely lost for recycling purposes. Recovered nutrients originate only from application of biowaste compost, amounting to 0.03 kg N, 0.12 kg P and 0.48 kg K per person and year. If sewage sludge is applied in agriculture (R_{agri}), these amounts rise to 0.43 kg N, 0.61 kg P, and 0.49 kg K per person and year (Figure 35). Sewage sludge contains only a part of wastewater N (~ 18% of influent N), but the majority of wastewater P (~ 96% of influent P). Nitrogen is mostly transferred to gaseous N_2 in denitrification and is thus lost for recycling purposes. Phosphorus is eliminated by chemical precipitation and transferred completely to sewage sludge. However, the plant availability of chemically-bound P is assumed to be limited (70%). Hence, the relative share of total nutrients in wastewater and biowaste which can be recovered by the agricultural application of sewage sludge amounts to 8% for nitrogen, 68% for phosphorus, and 27% for potassium (Figure 36).

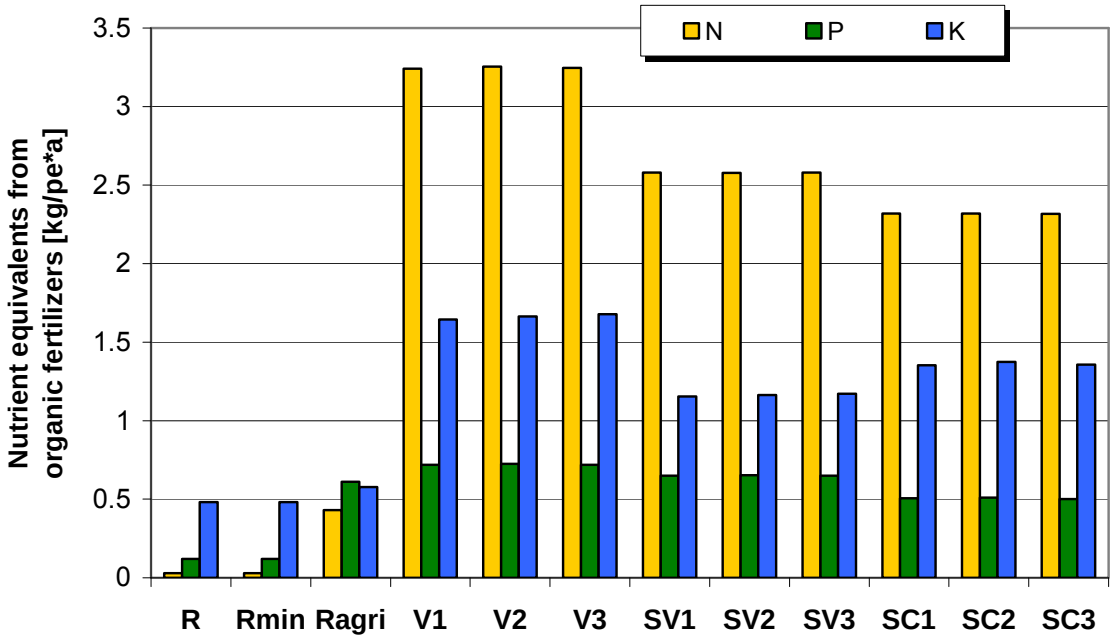

Figure 35: Nutrient equivalents (plant-available nutrients) provided by organic fertilizers

Separation scenarios

In the separation scenarios, nutrient recycling is one of the intended targets of wastewater treatment and disposal. Consequently, the amounts of recovered nutrients are comparable or higher than in conventional wastewater treatment, ranging between 2.32-3.25 kg N/(pe*a), 0.51-0.72 kg P/(pe*a), and 1.16-1.68 kg K/(pe*a). Surprisingly, the two-flow systems without urine separation are superior in nutrient recovery to the three-flow systems where urine is separately collected. This is due to the low efficiency of the separation toilets, where only 70% of the urine can be effectively collected. The remaining urine is lost for recycling purposes in the three-flow systems, either with the faeces filtrate (composting systems) or during the dewatering of the digester residual (digestion systems). In the two-flow systems where faeces and urine are collected together by vacuum and treated in digestion, no nutrient-rich urine is lost, because the digester residual is applied without dewatering. Even though the nitrogen emissions during application of organic fertilizers are assumed to be higher for the two-flow systems (23% of N) than for the three-flow systems (11% of urine-N, 6-7% of faeces-N), the two-flow systems are still superior in nitrogen recovery. The recovery ratio is highest in the two flow systems (scenarios V: 63/80/78% for N/P/K), smaller for the three flow systems with digestion (scenarios SV: 50/72/54), and smallest for the three-flow systems with composting (scenarios SC: 45/56/63).

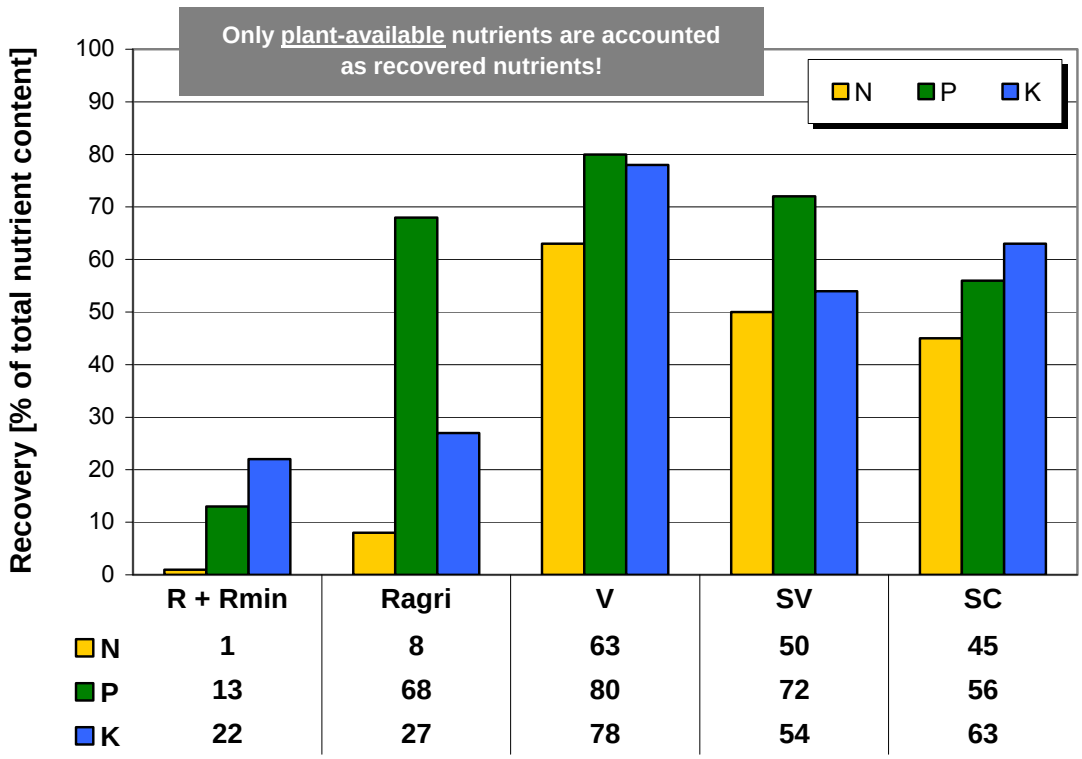

Figure 36: Recovery of nutrients in agriculture in relation to total amount of nutrients in wastewater and biowaste

The comparison of conventional and separation systems in terms of nutrient recovery potential is depending on the nutrient. For nitrogen and potassium, separation systems can significantly increase the recovery potential. For phosphorus, the application of sewage sludge in agriculture can lead to a comparable recovery potential between conventional and separation systems. The question of a possible limitation of the plant availability of P in sewage sludge is the crucial point here. Advantages for the separation systems result only from differences in plant availability of sewage sludge P (70%) and urine-derived P (100%).

In all, the comparison reveals a significant influence of the assumptions for a) plant availability of P in sewage sludge and b) the separation efficiency of the urine separation toilets. Hence, the influence of both parameters on the results is further investigated in sensitivity analysis.

5.1.3 Processes for system expansion

The secondary functions of separation systems are accounted for by expanding the scenarios with processes which deliver equivalent products (cf chapter 3.4). Two secondary functions are included in this study: the supply of electric energy and the supply of nutrients with organic fertilizers. The equivalent products are electric energy from the grid and industrial mineral fertilizer.

The calculation of the respective amount of equivalent products is based on the scenario delivering the highest amount of electricity and nutrients. Here, scenario V2 delivers maximum amounts for both secondary products. Consequently, all other scenarios are expanded in relation to scenario V2 (Table 62).

Electric energy

The amount of electric energy which has to be delivered by system expansion amounts to 3-48 kWh/(pe*a) depending on the scenario. Scenarios without energy recovery ($R/R_{min}/R_{agri}$ and SC1/2/3) naturally have to deliver the highest amounts of additional electricity with 36-48 kWh/(pe*a). Energy recovery scenarios are more or less comparable in their electricity output, so that these scenarios need only minor corrections via system expansion (3-7 kWh/(pe*a)). Scenario V2 has the maximum output of electricity as a secondary function and thus serves as the reference scenario for the calculation. In general, the secondary function of energy recovery in separation systems has a strong influence on the energetic comparison in this study.

Table 62: Electric energy and mineral fertilizer provided with system expansion processes

Scenario	Electric energy from the grid	Industrial mineral fertilizer		
		Nitrogen	Phosphorus	Potassium
	*[kWh/pe*a]*	*[kg/pe*a]*	*[kg/pe*a]*	*[kg/pe*a]*
R	35.9	3.2	0.6	1.2
R_{min}	47.7	3.2	0.6	1.2
R_{agri}	36.6	2.8	0.1	1.2
V1	3.5	0	0	0
V2	0	0	0	0
V3	3.4	0	0	0
SV1	6.7	0.7	0.1	0.5
SV2	3.2	0.7	0.1	0.5
SV3	6.5	0.7	0.1	0.5
SC1	47.9	0.9	0.2	0.3
SC2	47.2	0.9	0.2	0.3
SC3	47.5	0.9	0.2	0.3

scenario V2 delivers the maximum amount of energy and nutrients

Fertilizer

The supply of additional nutrients with mineral fertilizer amounts to 0.7-3.2 kg N, 0.1-0.6 kg P, and 0.3-1.2 kg K per person and year depending on the scenario. Total annual sales of mineral fertilizer in Germany (related to its total population) amount to 21.7 kg N, 1.5 kg P, and 4.3 kg K per person and year (DESTATIS, 2007). Thus, the proportion of the total demand for mineral fertilizer in Germany which can be substituted with organic fertilizers from wastewater and biowaste is relatively small for nitrogen (15%), but significant for potassium (28%) and especially phosphorus (40%). In other words, the recovery of nitrogen from wastewater and biowaste has only a minor impact on the total demand for mineral nitrogen fertilizer, whereas the recovery of potassium and phosphorus significantly lowers the demand for their mineral equivalents. This is especially important for phosphorus which is produced from phosphate rock, a finite resource which is depleted continuously (USGS, 2008).

5.1.4 Effluent concentrations and loads from wastewater and greywater treatment plants

The elimination of pollutants in the different treatment processes for wastewater and greywater is calculated by relative elimination ratios for the influent load of each substance (= % elimination) in the inventory. The relative elimination ratios for each process (activated sludge plant, soil filter, and membrane bioreactor) are estimated from characteristic performance data of wastewater treatment plants, based on nation-wide surveys or literature information of full-scale or pilot plants.

However, actual elimination ratios are typically within a certain range and depend on the specific wastewater composition, the load situation, and other specific features of the treatment plant. The exact prediction of average elimination ratios for a representative plant is difficult and thus connected to uncertainty, especially in quantifying the differences between the treatment of concentrated wastewater and greywater with low pollutant loads. To check the plausibility of the estimated elimination ratios of all wastewater treatment processes, average effluent concentrations are calculated for each scenario. These concentrations are compared between the scenarios and in relation to the legal discharge limits in Germany (AbwV, 2004). Mean effluent concentrations are calculated for each scenario in total by dividing the total effluent loads of all processes by the total effluent volume. For the reference scenarios, this refers only to the effluent from the wastewater treatment plant. For the separation scenarios, the considered processes include greywater treatment (eventually with faeces filtrate), but also the treatment of composting leachate or sludge liquor from digestion.

Chemical oxygen demand

The chemical oxygen demand is a sum parameter for the organic matter content of wastewater. All calculated effluent concentrations are between 31-53 mg/L COD and therefore comply with the legal discharge limit of 90 mg/L (Figure 37). Effluent concentrations of the reference scenarios are in the range of 47-52 mg/L COD, which are reasonable concentrations for the effluent of a conventional activated sludge plant treating concentrated wastewater without stormwater (DWA, 2005).

COD concentrations in the effluent of the greywater treatment plants are different between activated sludge plants (SBR, MBR = 32 mg/L COD) and soil filters (~ 50 mg/L COD). Due to the considerable evaporation of water in the soil filter (~ 10 L/pe*d), the reduced effluent volume leads to higher effluent concentrations in this calculation. In general, the low content of organic matter in greywater is expected to result in equal or lower effluent concentrations of COD compared to the reference systems.

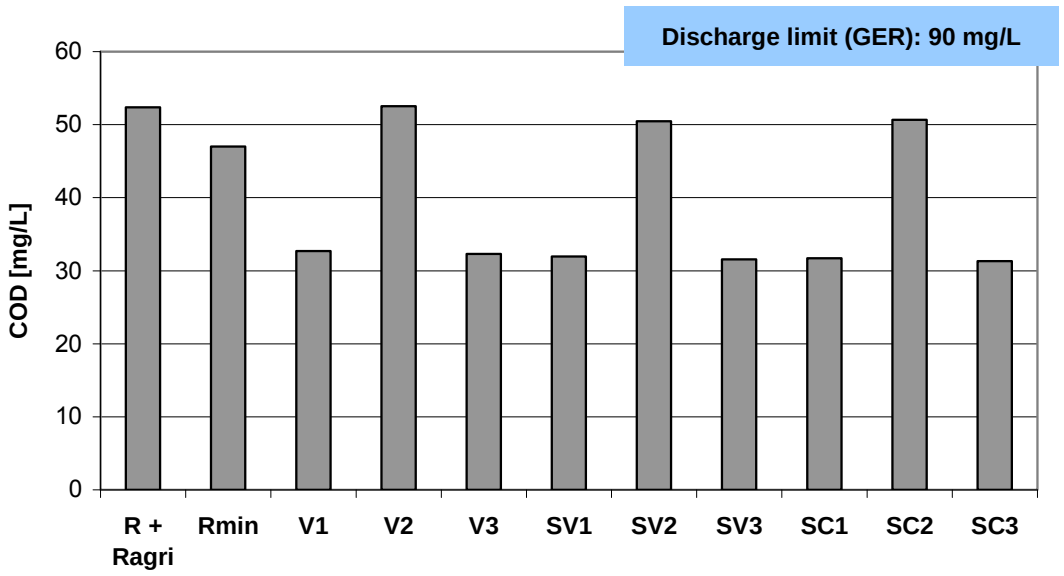

Figure 37: Calculated effluent concentrations for chemical oxygen demand

Nitrogen

Two parameters are regulated by law for the nitrogen content of the effluent: the ammonia nitrogen (NH_4-N) and the total inorganic nitrogen ($N_{total,\ inorg}$). Legal discharge limits are set at 10 mg/L NH_4-N (size category 3, > 5000 inhabitant equivalents) and 18 mg/L $N_{total,\ inorg}$ for larger plants (category 4, > 10000 inh.-eq). All scenarios safely comply with the limit for NH_4-N (Figure 38), and all but one comply with the standard for $N_{total,\ inorg}$.

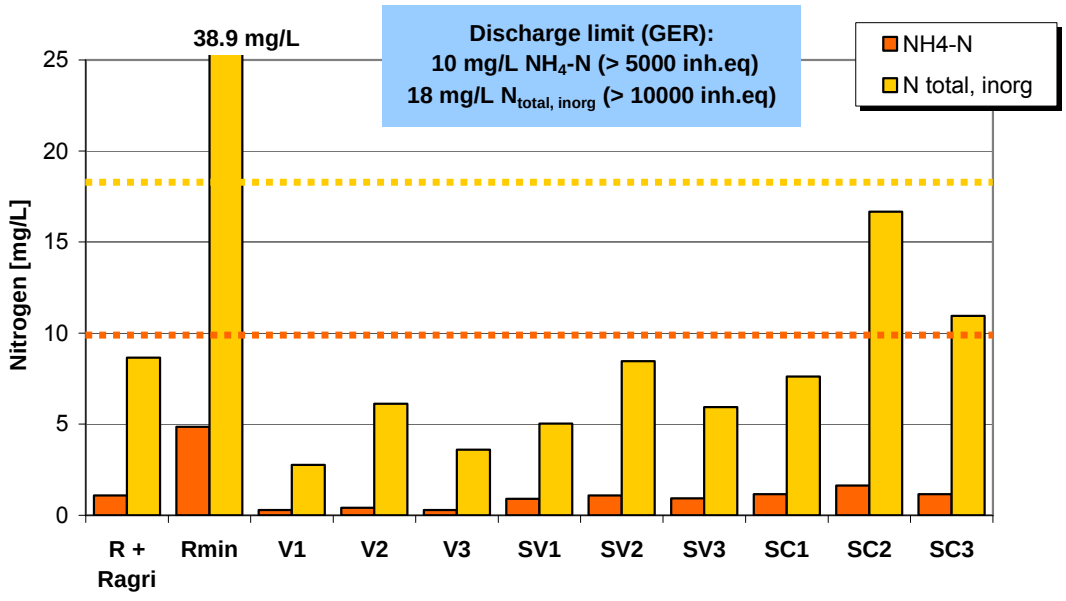

Figure 38: Calculated effluent concentrations for ammonia and total inorganic nitrogen

The reference system without extended nutrient removal (R_{min}) has the highest effluent concentration of 5 mg/L NH_4-N. The reference systems with advanced wastewater treatment (R + R_{agri}) and the separation systems with urine separation (SC + SV) are characterized by a comparable effluent concentration of 1 mg/L NH_4-N. In the separation systems, the co-treatment of concentrated faeces filtrate or sludge liquor from digestion (with concentrations of 100 and 315 mg/L NH_4-N, respectively) with greywater is responsible for the relatively high ammonia load in the effluent. The treatment of "pure" greywater with low influent concentration (~ 7 mg/L NH_4-N) results in effluent concentrations of 0.3 mg/L NH_4-N (scenarios V).

For the inorganic nitrogen, the denitrification capacity of the respective treatment process is decisive. The reference system without extended denitrification (R_{min}) has a high effluent concentration of 39 mg/L $N_{total,\ inorg}$, thus exceeding the limit for larger treatment plants (18 mg/L for plants > 10000 inh.-eq). However, the present study describes a system for 5000 inhabitants, so that this limit is not applicable here. It has to be kept in mind that the minimum reference system does not comply with the limits for effluent quality of larger plants. If the conventional system is equipped with denitrification (R + R_{agri}), the effluent has a concentration of 9 mg/L $N_{total,\ inorg}$.

In the separation scenarios, greywater treatment in soil filter leads to the highest effluent concentrations for inorganic nitrogen (6-17 mg/L) due to the low denitrification capacity (40%). MBR systems have a higher denitrification ratio (60% = 4-11 mg/L), followed by SBR plants (70% = 3-8 mg/L). Highest effluent concentrations are generally reached in the composting scenarios where concentrated faeces filtrate is treated together with greywater. In vacuum scenarios with dewatering of digester residual (SV), sludge liquor is treated separately at the biogas plant in an SBR process with high denitrification ratio (85%), leading to lower effluent concentrations of inorganic nitrogen. For treatment of "pure" greywater (scenarios V), effluent concentrations are between 3-6 mg/L $N_{total,\ inorg}$.

Phosphorus

A legal discharge limit for phosphorus only applies for larger treatment plants (2 mg/L for plants >10000 inh.-eq). Hence, it is not directly relevant for the scope of this study. All scenarios without extended phosphorus removal (R_{min} and soil filter scenarios SC2, SV2, V2) do not comply with this standard due to high predicted effluent concentrations of 3-9 mg/L P (Figure 39). In all other scenarios, chemical phosphorus removal via precipitation of ferric salts is applied. While the conventional activated sludge processes (R, R_{agri}, and SC1, SV1, V1) have comparable effluent P concentrations (0.6-0.8 mg/L), the treatment of greywater in a membrane bioreactor leads to lower effluent concentrations (0.3-0.4 mg/L in scenarios SC3, SV3, V3). This advantage of the MBR process is due to a better separation of particulate matter (flocculation of ferric iron) by

the membrane and is well-known for this type of treatment (Pinnekamp and Friedrich, 2006).

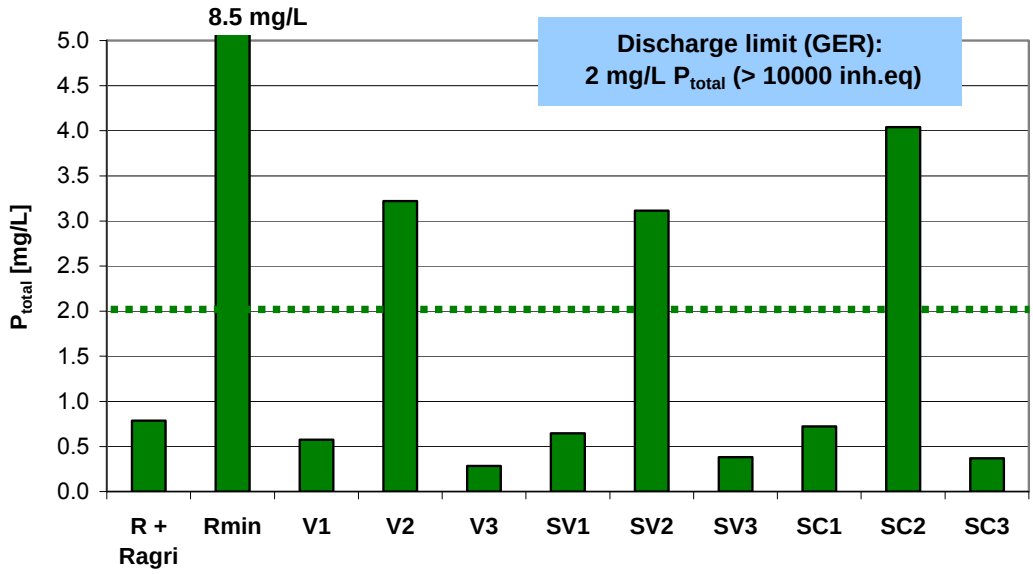

Figure 39: Calculated effluent concentrations for total phosphorus

Effluent loads

After the effluent concentrations have been checked for their plausibility, the effluent loads of the separation systems are compared to the reference system R with advanced wastewater treatment (Figure 40). These loads are the basis for the calculation of the eutrophication potential of each scenario, which is an important indicator in the impact assessment of wastewater treatment systems.

- **COD:** The COD load in the effluent can be reduced by 16-57% in separation scenarios. The separate treatment and recycling of organic matter (mostly faeces-derived) relieves the treatment process and should lead to lower effluent loads.

- **Nitrogen:** For nitrogen, the comparison is inconclusive (Figure 40). Composting scenarios may increase the effluent load of N_{total} compared to conventional wastewater treatment. The nitrogen load of concentrated faeces filtrate treated together with greywater increases effluent loads for N_{total} by 7% in SBR scenario (SC1) and 22% in MBR scenario (SC3). For the soil filter scenario (SC2) with low denitrification, this increase is even higher and amounts to 114%. It is therefore questionable if the soil filter is an adequate treatment process for greywater mixed with faeces filtrate. Vacuum scenarios (SV +V) do not produce faeces filtrate due to the processing of the complete toilet wastewater in the digestor. Consequently, these scenarios decrease the effluent loads for N_{total} by 12-70% compared to the conventional system. The

highest decrease (- 39-70%) can be reached in the vacuum scenarios V where the digester residual is applied without dewatering (= no sludge liquor).

- **Phosphorus:** The comparison reveals that soil filters are not comparable to advanced wastewater treatment in terms of P removal. Even though the separate treatment of toilet wastewater leads to relatively low phosphorus loads to the soil filters, the total P effluent loads increase by 170-345% in soil filter scenarios SC2/SV2/V2 compared to the reference system R. The low P removal capacity of the soil filter systems (50%) is a major disadvantage compared to advanced wastewater treatment with chemical P elimination (>95%). This drawback is especially significant in case of co-treatment of faeces filtrate with greywater (SC2), but also if "pure" greywater with low phosphorus content is treated in soil filters (V2 and SV2).
  All separation systems with activated sludge technology (SBR or MBR) and chemical P elimination decrease P effluent loads by 11-75% compared to the reference system R. The highest reductions are reached with MBR systems due to their superior particle retention capacity (- 63-75%).

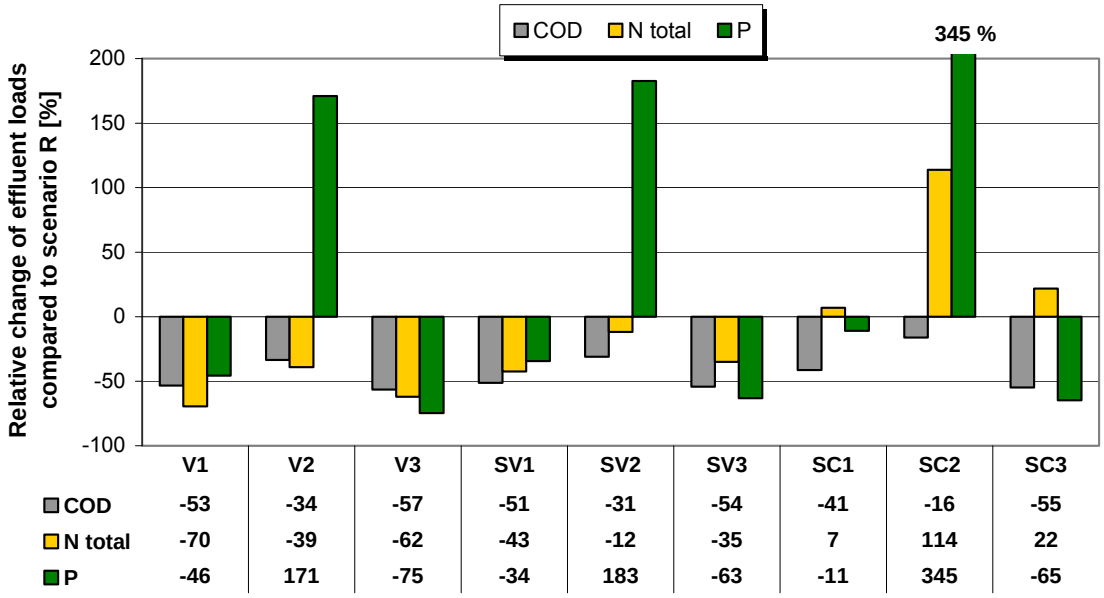

Figure 40: Variation of effluent loads in separation scenarios compared to wastewater treatment with extended nutrient removal (scenario R)

In all, it can be concluded that separation systems can offer potentials for a decrease of pollutant loads in the effluent. However, a careful examination of the system configuration is required to realize these potentials:

- Soil filter systems typically provide only limited nutrient elimination. These natural systems can be inferior to advanced wastewater treatment with

extended nutrient removal, especially concerning long-term phosphorus removal. Thus, total nutrient loads in the effluent may increase in soil filter scenarios compared to the reference system, even though nutrient-rich toilet wastewater is treated separately.

- The treatment of high-strength flows (e.g. faeces filtrate) in a soil filter leads to very high effluent loads for N and P and eventually exceeds the legal effluent standards for larger plants.
- Greywater treatment in an activated sludge process (SBR or MBR) results most likely in a decrease in pollutant loads compared to the reference systems. The MBR process is characterized by slightly higher nitrogen loads in the effluent, but has a superior P removal capacity due to the excellent particle separation with the membrane.

Further remarks

The above calculation of effluent loads and concentrations is based on estimations for the performance of the different treatment processes under varying conditions (influent concentrations, load profiles etc). More data from pilot and full-scale plants is required to confirm the reduction potentials of separation systems. The estimation of representative elimination ratios for pollutant removal from literature and plant data determines the effluent loads in each scenario and consequently has a strong influence on the respective environmental impact indicator for eutrophication.

In sensitivity analysis, the influence of an alternative approach to this issue is investigated: it is assumed that all scenarios with comparable technology produce the same effluent quality (= pollutant concentrations). Thus, all activated sludge plants with extended nutrient removal (scenarios R + R_{agri} and SC1/SV1/V1) are characterized by the same effluent concentrations for COD, N, and P regardless of the different influent loads. The advantage of superior particle retention of MBR systems is maintained in this case, as is the limited nutrient elimination of soil filters.

5.1.5 Heavy metals loads emitted to surface waters and soil

The emission of heavy metals into the environment is an important factor determining the ecotoxicological impacts of the sanitation systems. The heavy metal content of wastewater constituents can be transferred to receiving surface waters with the effluents or to agricultural soil with organic fertilizers. In case of sewage sludge incineration, the heavy metals bound in the sludge are disposed in a landfill or e.g. in road construction and are not accounted as emissions into the environment. Mineral fertilizer as an equivalence product in system expansion is also contaminated with heavy metals and contributes to the loads emitted to agricultural soil.

Heavy metal emissions to surface waters
Heavy metal emissions to surface waters predominantly occur via the effluent of the wastewater treatment process. Copper and zinc are selected to exemplify the differences between conventional and separation systems:

- Emissions of zinc are lower in all separation scenarios compared to the reference system (Figure 41). Emissions of copper are lower in separation scenarios with activated sludge for greywater treatment, whereas soil filter scenarios lead to a small increase in Cu effluent loads.
- Elimination ratios for these metals are comparable between wastewater and greywater treatment (activated sludge plants: 85% for Cu, 75% for Zn; soil filters: 80%/80%).
- Copper and zinc mainly originate from elevated concentrations in drinking water (0.16 mg/L Cu and 0.37 mg/L Zn) due to corrosion of pipe materials. Consequently, reduced water consumption with low-flush and vacuum toilets decreases input loads of these metals and results in fewer emissions to surface waters. Similarly, greywater reuse lowers Cu and Zn input by substituting drinking water.
- The separate treatment of faecal matter transfers the faecal-derived Cu and Zn to organic fertilizer and not to wastewater treatment.

For other heavy metals, relevant inputs originate mainly from greywater composition and to a smaller extent from drinking water and faeces, but also from the production of mineral P fertilizer (e.g. Cd, Cr). Hence, total effluent loads are slightly reduced due to substitution of mineral P fertilizer, reduction of drinking water consumption and separate treatment of faeces (e.g. reduction of Pb: 5–23%, Cr: 12-31%, Cd: 19-36%, cf. annex 12.9). In general, more data from pilot and full-scale plants is required to exactly predict heavy metal removal in greywater treatment. Currently, relative elimination is assumed to be comparable for wastewater and greywater treatment processes.

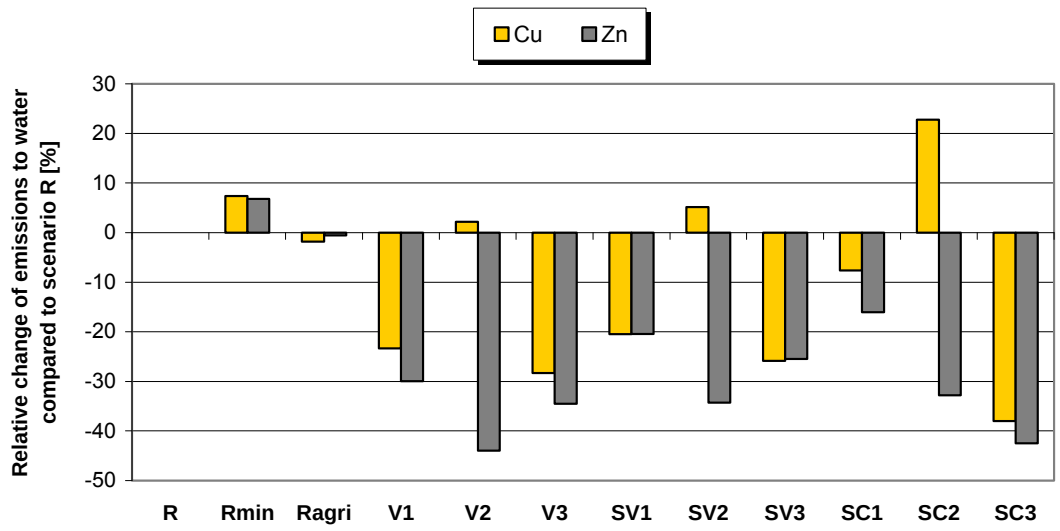

Figure 41: Emissions of Cu and Zn to surface waters (relative to scenario R)

Heavy metal emissions to agricultural soil

Input of heavy metals into agricultural soil occurs via organic or mineral fertilizers. For the essential trace elements of copper and zinc, the substitution of mineral with organic fertilizers in separation systems leads to an increase in the loads to agricultural soils (Figure 42). This corresponds to the lower emissions of Cu and Zn to surface waters: these metals are partially diverted from the water path to the soil in separation systems. Furthermore, concentrations of these metals in organic fertilizers from faeces and biowaste are higher than in mineral fertilizers. The application of sewage sludge in agriculture even extends this effect: copper loads to agricultural soil increase by 825% and zinc loads by 260% in scenario R_{agri}.

For other heavy metals, the application of organic fertilizers in separation systems leads to a significant reduction (Figure 43). Loads to agricultural soil decrease by 48-76% for Cd, 52-78% for Cr, 31-48% for Pb, 11-47% for Ni, and 65-100% for U compared to the application of mineral fertilizer (scenario R). Organic fertilizers from urine, faeces and biowaste have a low content of these heavy metals compared to an average mineral fertilizer. The quality of sewage sludge from conventional wastewater treatment is even worse than mineral fertilizer: in scenario R_{agri}, agricultural application of sewage sludge increases metal loads to soil by 14% (Cd), 22% (Cr), 120% (Ni), 141% (Hg), and 145% (Pb). In general, sewage sludge is characterized by a high content of heavy metals compared to organic or mineral fertilizers. An exemption is the radioactive element uranium, which is only contained in mineral phosphate fertilizer.

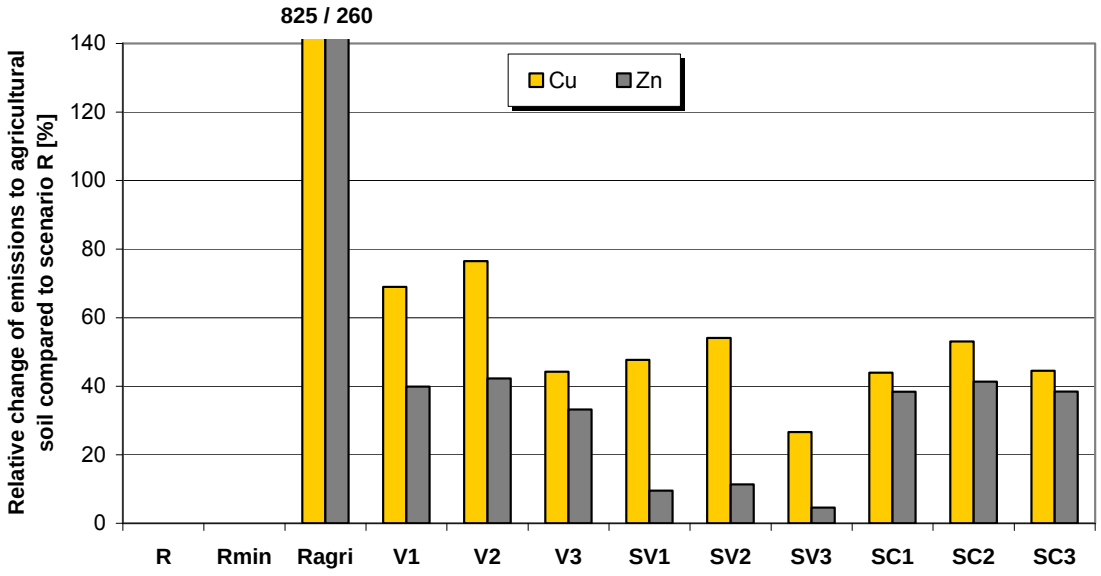

Figure 42: Emissions of Cu and Zn to agricultural soil (relative to scenario R)

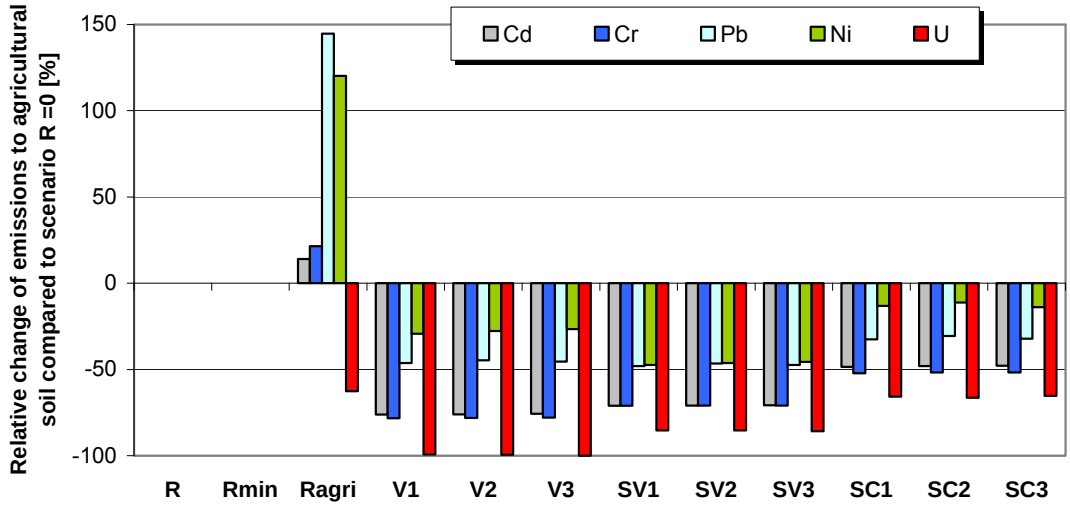

Figure 43: Emissions of Cd, Cr, Pb, Ni, and U to agricultural soil (relative to scenario R)

Overall, it becomes apparent that organic fertilizers from urine, faeces and biowaste are less contaminated with toxic heavy metals Cd/Cr/Ni/Hg/Pb than mineral fertilizer or sewage sludge. For Cu and Zn, organic fertilizers have a higher content than mineral fertilizer, but significantly less than sewage sludge. The substitution of mineral P fertilizer with organic fertilizer from secondary resources leads to a significant reduction in heavy metal loads to agricultural soil. If the nutrients are collected in a separation system, they are not contaminated with heavy metals from other wastewater sources. Thus, organic fertilizers from separation systems have a superior quality for agricultural application compared to sewage sludge from the treatment of combined wastewater.

Quality of sewage sludge

The quality of the sewage sludge calculated in this study is compared to mean values of sewage sludge applied in agriculture in Germany and to legal limits for agricultural application of sewage sludge (Table 63). The calculated quality of the sewage sludge in scenario R_{agri} is higher than mean values of sewage sludge applied in agriculture in Germany. However, they are still safely within the current legal limits for sewage sludge application established in 1992 (AbfKlärV, 1992). This comparison confirms that the quality of the sewage sludge calculated in this study is a realistic estimation of typical sewage sludge in Germany.

The low heavy metal content of the sludge which is effectively applied to agriculture in Germany illustrates the efforts to improve the quality of the sludge. By limiting industrial heavy metal emissions to the combined sewer, heavy metal content of municipal sewage sludge has been decreasing continuously over the last decades (BMU, 2007). This improvement should facilitate the safe disposal of sewage sludge in agriculture, especially concerning the public reception of this disposal route. Otherwise, sludge disposal can be costly, e.g. through fees for incineration or landfill deposition.

However, if lower limits for heavy metal contamination will be established in the future (BMU, 2006), the proportion of sewage sludge that can be disposed in agriculture may be heavily restricted by its heavy metal content. In other European countries (CH, NL), the agricultural application of sewage sludge is completely forbidden or heavily restricted. The problematic risk assessment between benefits and drawbacks of agricultural disposal of sewage sludge could be overcome with the use of separation systems delivering an organic fertilizer with low content of inorganic and organic pollutants.

Table 63: Comparison of calculated heavy metal loads in sewage sludge with German mean values and legal limits

Element	This study	Mean values for Germany 2006	Limits for sewage sludge application
[mg/kg dry matter]	*scenario R_{agri}*	[1]	[2] ([3])
Cd	4	1	10 (2)
Cr	54	37	900 (80)
Cu	461	300	800 (600)
Ni	30	25	200 (60)
Hg	0.7	0.6	8 (1.4)
Pb	54	37	900 (100)
Zn	1043	714	2500 (1500)

1) for sewage sludge used in agriculture (BMU, 2007)
2) AbfKlärV, 1992
3) proposal of new limits for amendment of AbfKlärV (BMU, 2006)

5.2 Results of the Life Cycle Impact Assessment

In Life Cycle Impact Assessment, the results of the Life Cycle Inventory are transferred into indicator results via classification and characterization (cf chapter 3.8). Here, all calculated indicator results are presented and analysed by contribution analysis, thus determining the processes or emissions which contribute most to the indicator category.

5.2.1 Cumulative energy demand

The cumulative energy demand (CED) of non-renewable resources (fossil and nuclear fuels) accounts to 750 to 1430 MJ per person and year for all scenarios (Figure 44). The reference systems require 1200 to 1430 MJ/(pe*a) depending on whether energy recovery via sewage sludge digestion is applied ($R + R_{agri}$) or not (R_{min}). Major parts are contributed by the operation of the systems (~ 45%) and the supply of equivalence products (~ 50%), whereas the infrastructure constitutes only a small part of the total CED (6-8%). It is remarkable that the CED for the secondary functions (fertilizer and energy supply) is in the same range as CED for operation and infrastructure in the reference systems.

For the separation systems, the total CED ranges between 750 and 1430 MJ/(pe*a) depending on the system configuration. As expected, separation systems with energy recovery are superior to systems with faeces composting. The recovery of energy bound in the organic matter of faeces and biowaste outweighs the increased operational energy demand of vacuum systems. Consequently, the CED for the supply of equivalence products (= energy) is significantly higher in composting systems.

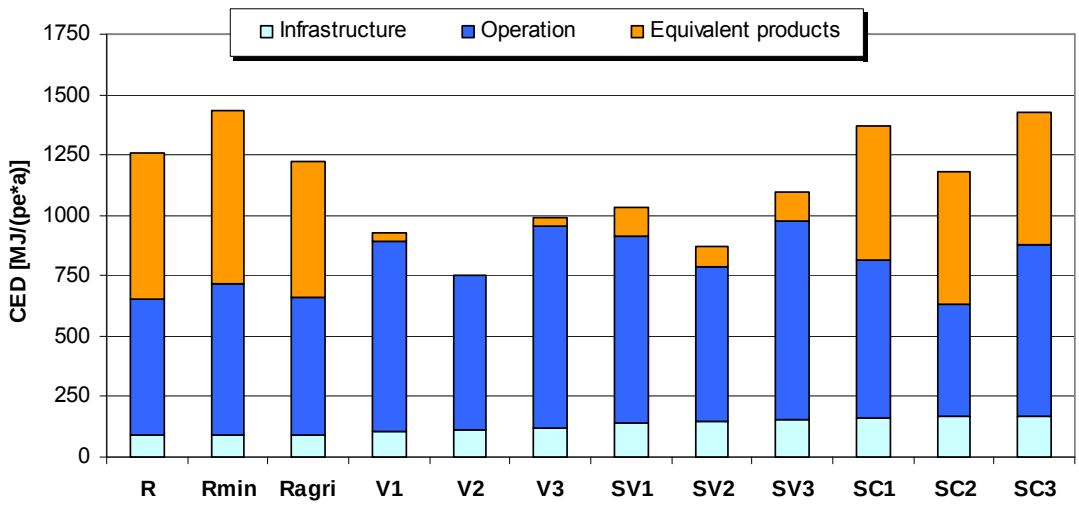

Figure 44: Cumulative energy demand

Greywater treatment is another important factor for the CED of separation systems. In each scenario group, the second scenario with a low-energy process (soil filter) has a significantly smaller CED than the first and third scenarios with technical processes (SBR or MBR). The reuse of greywater for toilet flushing does not result in an energetic benefit. The amounts of substituted drinking water are too small (5 L/(pe*d) in case of vacuum toilets and 24 L/(pe*d) for low-flush gravity toilets) to equalize the increased energy demand of the membrane system.

A detailed contribution analysis for the CED of infrastructure, operation and the supply of equivalence products is carried out to provide further insights in the energetic comparison.

Infrastructure

The cumulative energy demand for the construction of a conventional sanitation infrastructure with combined sewer is around 90 MJ per person and year (Figure 45). More than 50% are due to the construction of the sewer, while in-house piping and sewage treatment plant each require around 25% of total CED for infrastructure.

Separation systems are characterized by multiple flows, which need multiple piping networks for their transport. Consequently, the energy demand for the infrastructure is increased by 10-30% for two-flow systems (black-/greywater: V1-3) and 55-90% for three-flow systems (urine/brown-/greywater, urine: SC1-3 + SV1-3) compared to the reference scenario. This increase is in a large part caused by the additional sewer pipes (Figure 45). In case of greywater reuse, the required extra pipe network for service water supply needs additional energy.

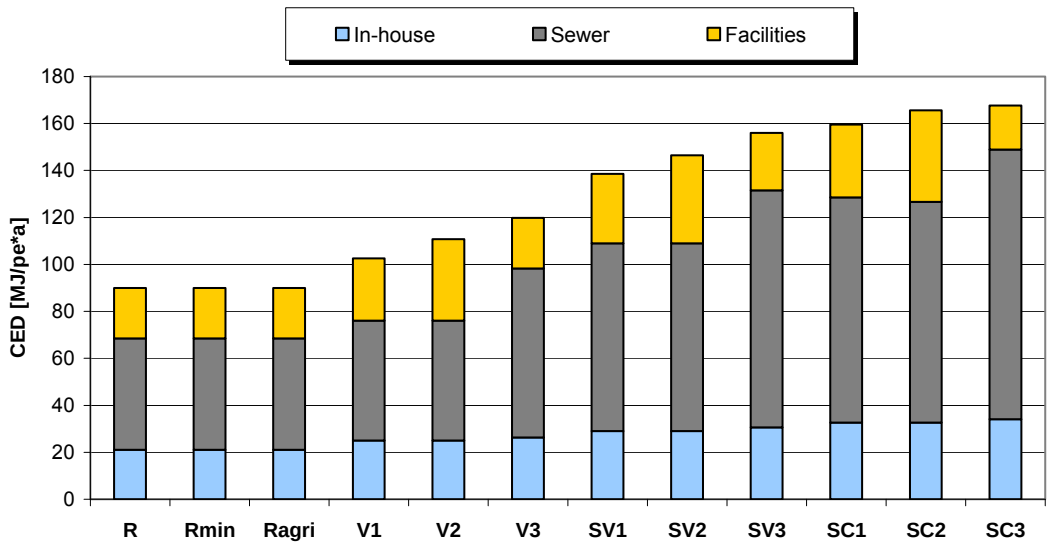

Figure 45: Cumulative energy demand for infrastructure

In all, the separation scenarios have a higher CED for their infrastructure due to the additional piping. However, the increase in relation to the conventional system is less than a factor of 2 or 3, which could be expected by their layout (two-flow or three-flow systems vs one combined sewer). The small diameter of the additional sewer pipes (e.g. for vacuum system or urine transport) and the installation of all pipes in one trench minimizes the extra energy demand of separation systems to 10-90%.

Operation

The operational energy demand is in the range of 560-620 MJ/(pe*a) for the conventional systems and 470-840 MJ/(pe*a) for the separation scenarios. In relation to the respective CED for infrastructure, the operational CED is higher by a factor of 6-7 for conventional systems and 3-8 for separation systems.

The contribution analysis reveals that the operational CED is largely dominated by the electricity supply (Figure 46). This process contributes 75-90% to the total operational CED, whereas the contribution of chemical supply (e.g. flocculants) and transport are small. Only in vacuum scenarios without urine separation (V1-3), the high volume of the organic fertilizer (= digester sludge, not dewatered) leads to a significant contribution of transport to the operational CED (15-19%).

In general, separation scenarios have a higher operational CED (+13-47%) than the conventional systems, particularly due to higher electricity demand (cf. chapter 5.1.1) and to a lesser extent due to increased transport volumes. An exemption is scenario SC2 with faeces composting and soil filter, which has the lowest operational CED of all scenarios (467 MJ/pe*a).

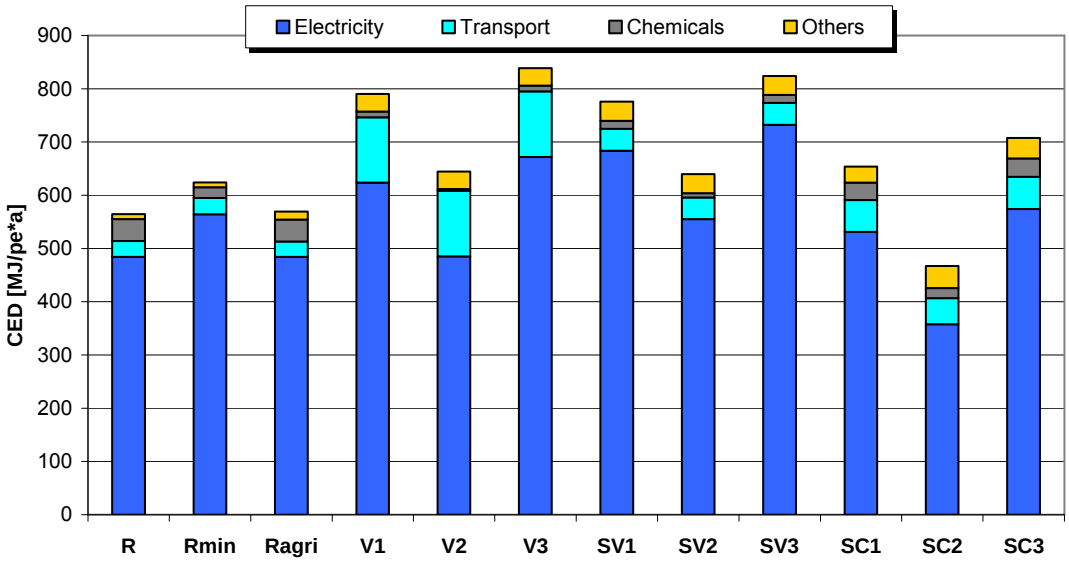

Figure 46: Cumulative energy demand for operation

Equivalent products

For the supply of equivalent products (grid energy and mineral fertilizer), the CED is highest for scenario R_{min} (720 MJ/pe*a) without nutrient recycling or energy recovery of sewage sludge (Figure 47). Energy recovery via sewage sludge digestion decreases the CED for equivalent products substantially to 600 MJ/(pe*a) in scenario R. If sewage sludge is applied in agriculture for nutrient recycling, this amount further decreases slightly to 561 MJ/(pe*a) in scenario R_{agri}. The energetic benefits of nutrient recycling via sewage sludge are limited due to the low content of nitrogen in the sludge and its low plant availability (50%). The production of mineral nitrogen fertilizer is by far more energy intensive than mineral phosphate fertilizer. Hence, the high ratio of phosphorus recycling with sewage sludge (cf. chapter 5.1.2) results only in a small energetic benefit (< 40 MJ/(pe*a)).

For the separation systems, the option of energy recovery from organic matter is crucial for the comparison with the reference system. Composting systems without energy recovery are comparable to the reference scenarios, whereas vacuum systems with energy recovery offer substantial benefits. Two-flow vacuum systems without urine separation have the highest energy recovery (including energy bound in organic matter of urine) and supply the most organic nitrogen fertilizer. Consequently, these scenarios have to supply the smallest amount of equivalent products.

While comparing the different separation systems, the amount of substituted mineral fertilizer plays only a minor role, whereas the energy recovery option is again crucial for this comparison (Figure 47).

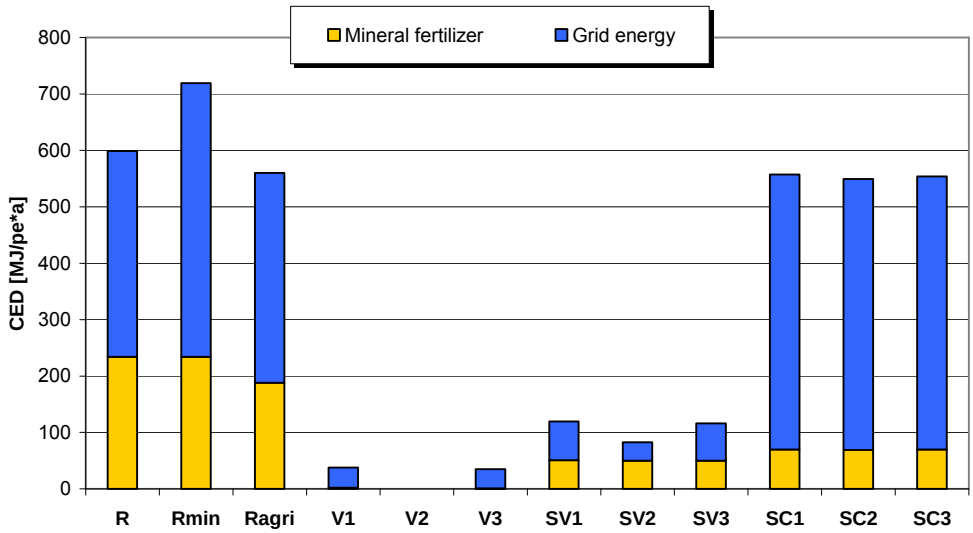

Figure 47: Cumulative energy demand for supply of equivalent products (system expansion)

In general, the following conclusions can be drawn from the contribution analysis of cumulative energy demand:

- The supply of equivalent products has a decisive impact on the energetic comparison of conventional and separation systems.
- The choice of the greywater treatment process is equally important. Natural low-energy treatment systems (soil filter) result in substantial energetic benefits.
- Separation systems require more energy for the construction of infrastructure, but this is not decisive for the overall comparison due to its low contribution to the total CED (11-17%).
- Energy recovery via conversion of organic matter into biogas offers significant net energetic benefits only if biowaste is co-digested with toilet wastewater.
- In this study, urine separation (group SV) does not offer an energetic advantage compared to vacuum systems (group V) treating blackwater (= faeces and urine). By applying the digested blackwater directly to agriculture without dewatering, more nitrogen is supplied to the fields than with urine separation. In contrast, nutrients in urine which is not properly separated in separation toilets (30% of total urine) are mainly lost for recycling purposes. It has to be noted though that urine separation allows the partial containment of micropollutants via ozonation of separated urine. Direct application of digested blackwater does not provide a feasible option for micropollutant removal.
- Composting systems have the lowest energy demand for operation, but no option for energy recovery. Vacuum systems are energetically superior to composting systems.
- Transports have only a negligible impact on the overall comparison at the estimated distances for the transport of organic fertilizers (20 km).
- The reuse of greywater for toilet flushing does not lead to energetic benefits due to the low volume of substituted drinking water (5-24L/pe*d).

Hence, separation systems do not necessarily lead to energetic benefits compared to an optimized conventional system. If both the substitution of nitrogen fertilizer and the recovery of energy from the organic matter are implemented, the higher operational energy demand of separation systems is offset and substantial energetic benefits can be realized. Additional benefits can be realized if low-polluted greywater is treated in low-energy natural systems. The maximum reduction potential can be realized in scenario V2 if vacuum systems with energy recovery are combined with greywater treatment in soil filters (-40% compared to reference system).

Many system parameters of the Life Cycle Inventory have a significant influence on the cumulative energy demand. System configurations that were initially expected to yield an energetic benefit (urine separation, greywater reuse) may do so if certain system

parameters are optimized or assumptions are changed. For sensitivity analysis, the following parameters are chosen to determine their influence on the energetic comparison:

- Addition of biowaste in co-digestion
- Separation efficiency of urine separation toilets
- Energy demand for urine treatment and vacuum plant
- Transport distances for organic fertilizers
- Volume of reused greywater

5.2.2 Depletion of abiotic resources

The depletion of abiotic resources evaluates the demand for non-renewable energetic resources (fossil and nuclear fuels) as well as mineral resources (Fe, Al, Cu, P etc). In this study, the calculated indicator results are predominantly determined by the energy resources. The contribution of minerals to the abiotic depletion potential (ADP) is negligible for all scenarios, accounting for less than 0.1% of total ADP. Scenarios with energy recovery have benefits compared to the reference system, while composting scenarios are comparable or slightly worse than the conventional system (Figure 48). The benefit from recycling of wastewater-derived P as a substitution of raw phosphate being a limited resource is not reflected in ADP. Characterization factors for ADP of raw phosphate are too low to show this effect in the present indicator category. The limited availability of raw phosphate (demand vs projected reserves) seems to be negligible compared to the limited reserve and high demand of fossil fuels.

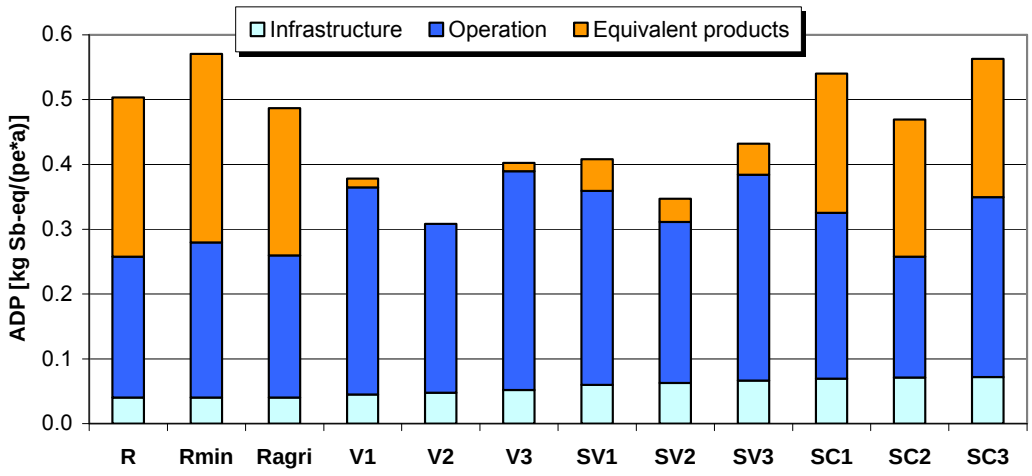

Figure 48: Abiotic depletion potential

5.2.3 Global warming

Typically, the indicator of global warming potential (GWP) is strongly related to the fossil energy demand due to CO_2 emissions from the burning of fossil fuels. However, emissions of CH_4 and N_2O can also have a strong influence on GWP, especially for agricultural processes.

In this study, the global warming potential is in the range of 75-150 kg CO_2-eq per person and year for all scenarios (Figure 49). The construction of the infrastructure constitutes only a minor part of the total GWP (5-13%), whereas system operation and the supply of equivalent products cause the major part of GWP. Similar to the cumulative energy demand, separation systems perform worse compared to the reference system for the GWP caused by infrastructure (+14-63%) and system operation (+12-45%). However, GWP from the supply of equivalent products offsets these amounts and results in an overall benefit for most separation scenarios. Vacuum scenarios reduce total GWP by 24-46% , while composting scenarios are comparable to the reference system.

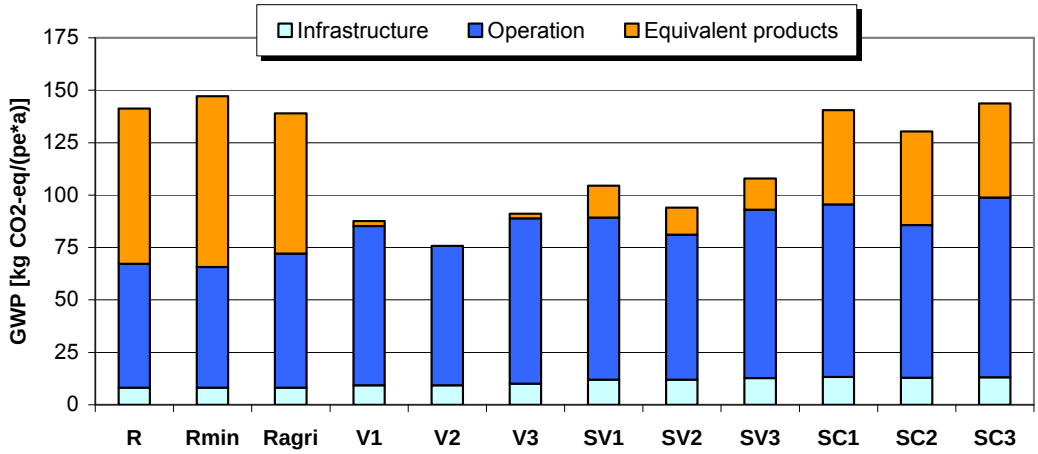

Figure 49: Global warming potential

Composting systems have a comparable GWP with the reference system, only the soil filter scenario has a slight benefit (-8%). The differences between the scenarios can be partially explained by the results from the energetic comparison (cf. chapter 5.2.1): separation systems require more energy for infrastructure and system operation, but deliver secondary products and thus avoid emissions involved in their production. Additionally, a contribution analysis for the equivalent products reveals that the substitution of mineral fertilizer plays a more significant role for the benefits in GWP than the supply of additional energy (Figure 50). Two factors are important here: a) the substitution of mineral nitrogen fertilizer whose production is energy-intensive and b) emissions of N_2O during the production of mineral N fertilizer.

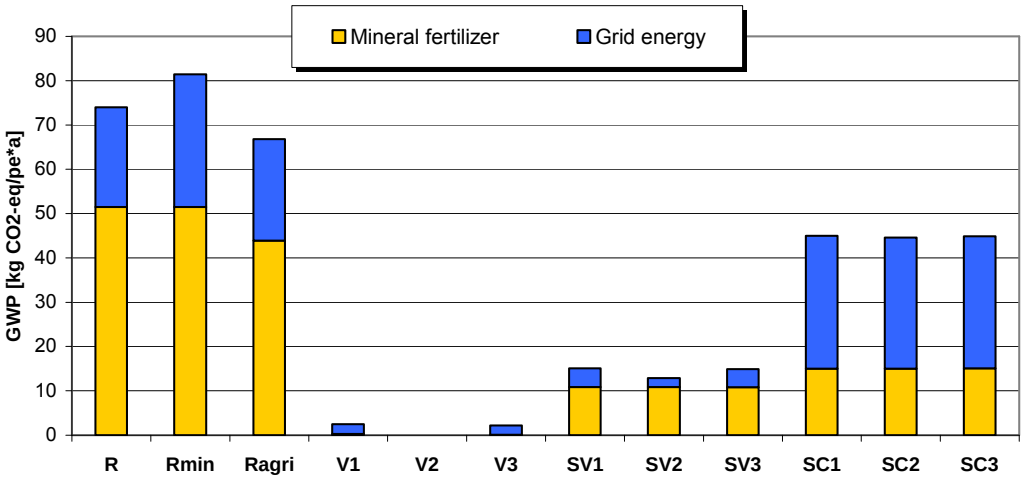

Figure 50: Global warming potential for supply of equivalent products (grid energy and mineral fertilizer)

Apparently, both CO_2 and N_2O emissions are important to assess the global warming potential of sanitation systems. Hence, a detailed contribution analysis of the different greenhouse gases is calculated to point out the decisive factors (Figure 51).

CO_2

Emissions of CO_2 from fossil sources mainly derive from energy production and transport processes. Consequently, the amount of CO_2 emissions is strongly correlated with the cumulative energy demand. In the contribution analysis for gases, the vacuum scenarios with low CED reduce CO_2 emissions in relation to the conventional system (Figure 51). Scenarios SV3 and SC1/2 have emissions of CO_2 that are comparable to the conventional system, whereas scenario SC3 increases CO_2 emissions. In general, benefits in CO_2 emissions do not seem to play a decisive role for the benefits in GWP of separation systems.

In all, the contribution of CO_2 is determined by the demand for electric energy in most scenarios. The contribution of transport processes to CO_2 emissions is negligible in most scenarios (<1% of CO_2 emissions). The maximum contribution of transport is calculated for the scenario with low electric energy demand and high transport volumes (scenario V2: 15% contribution of transport to CO_2 emissions, data not shown).

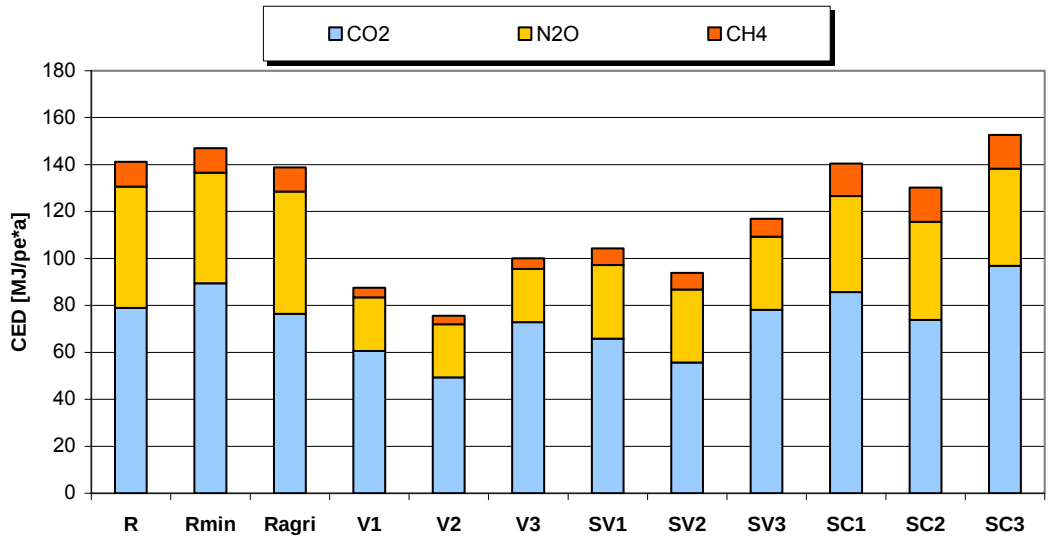

Figure 51: Greenhouse gases contributing to global warming

N_2O

Emissions of N_2O mainly arise during the production and application of nitrogen fertilizers and from the denitrification process in wastewater treatment:

- In the inventory, N_2O emission factors for the application of nitrogen fertilizers are assumed to be comparable for both mineral and organic fertilizers, so fertilizer application is not responsible for the benefits of separation systems.
- N_2O emissions during the production of mineral N fertilizer are substantial (~ 30% of total N_2O emissions in reference scenarios) and can be avoided in separation scenarios. In composting scenarios, N_2O emissions from composting partially offset this benefit.
- N_2O emissions from denitrification (~ 18% of total N_2O in reference scenarios) are reduced considerably due to lower influent loads of nitrogen and in greywater treatment.

Hence, two decisive factors have been revealed: a) avoiding N_2O emissions during fertilizer production and b) less N_2O emissions from denitrification. In total, composting scenarios reduce N_2O emissions by ~ 20% and vacuum scenarios by 40-56% compared to the reference system (Figure 51). This reduction of N_2O emissions is mainly responsible for the reduction of total GWP.

CH_4

Methane is mainly emitted in composting and to a smaller extent in energy production. Consequently, composting scenarios increase CH_4 emissions by 34-37% compared to the reference system, while vacuum scenarios reduce CH_4 emissions by 25-65%.

However, the contribution of CH_4 to the total GWP is small (4-12%) and it is not decisive for the overall comparison.

5.2.4 Acidification

The acidification potential (AP) is determined by the emissions of acidifying gases (mainly NH_3, NO_x, and SO_2). In this study, NH_3 emissions are responsible for the greatest share (70-89%) of the acidification potential in all scenarios due to high emissions from nitrogen fertilizer application and composting. Emissions of NO_x and SO_2 which typically occur in combustion processes are less important in this study (6-16% and 5-14%, respectively).

Overall, the acidification potential is between 0.5 and 1.2 kg SO_2-equvialents per person and year (Figure 52). Separation systems increase AP by 59-111% compared to the reference system. The substantial increase originates from higher emissions during system operation, whereas the infrastructure has a negligible impact on this indicator.

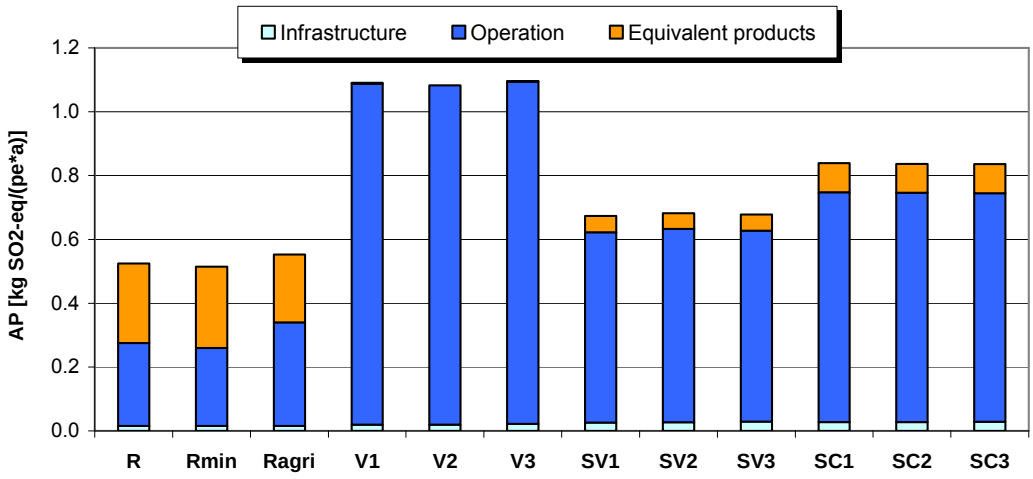

Figure 52: Acidification potential

By calculating the contribution of different sub-processes to acidification (Figure 53), the following conclusions can be drawn:

- The acidification potential is predominantly determined by the processes of fertilizer application and composting.
- The application of organic fertilizers is presumably associated with high NH_3 emissions. The respective NH_3 emission factors are estimated to be considerably higher for urine (10% of applied N) and digester sludge (22%) than for mineral fertilizer (5%).

- Composting processes also contribute substantially to NH_3 emissions, especially if faeces are co-treated with biowaste. Emissions originate mainly from open composting without encapsulation and off-gas cleaning.
- Emissions from other processes (transports, energy production etc) are comparable between all scenarios and play only a minor role for acidification (9-21% of total AP).

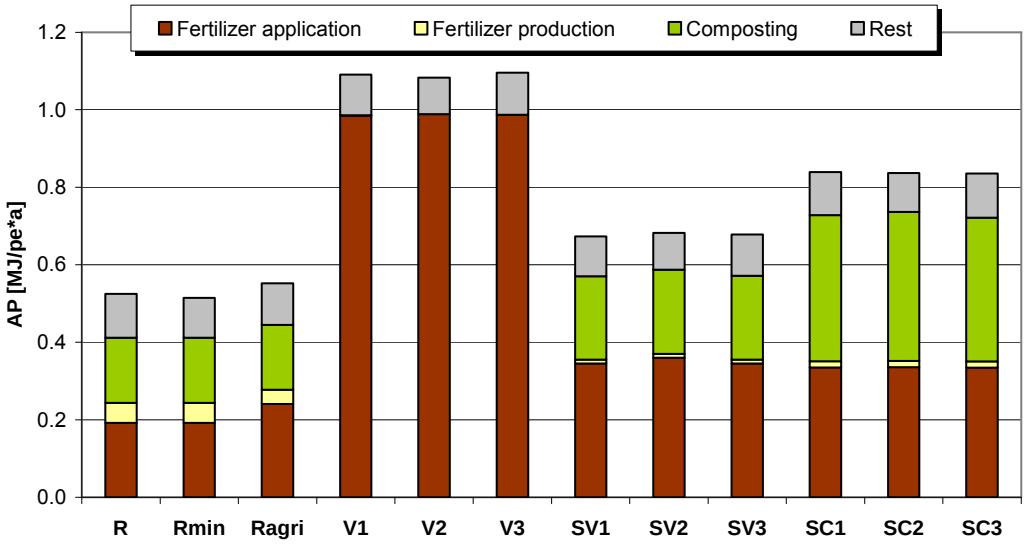

Figure 53: Contribution of sub-processes to acidification potential

In all, the substitution of mineral with organic fertilizers leads to a substantial increase in NH_3 emissions and thus in the acidification potential. Therefore, available emission reduction measures should be implemented to minimize this drawback for the separation systems. These measures include special application techniques for urine and digester sludge (e.g. direct injection in the ground, drag hoses, instant ploughing for incorporation of applied fertilizer) and encapsulation of the complete composting process with adequate off-gas cleaning in a biofilter. The effect of emission reduction measures for fertilizer application and composting on the acidification potential is quantified in sensitivity analysis.

5.2.5 Eutrophication

Eutrophication is mainly caused by the emission of nitrogen, phosphorus, and organic matter (as COD) to surface waters. Additionally, atmospheric deposition of nitrogen gases (NH_3, NO_x) contributes to this impact category to a lesser extent. Consequently, the decisive process for eutrophication is the treatment of wastewater and the associated effluent loads of nutrients. The contribution of infrastructure and equivalent products is negligible for the comparison of the eutrophication potential (EP) (Figure 54).

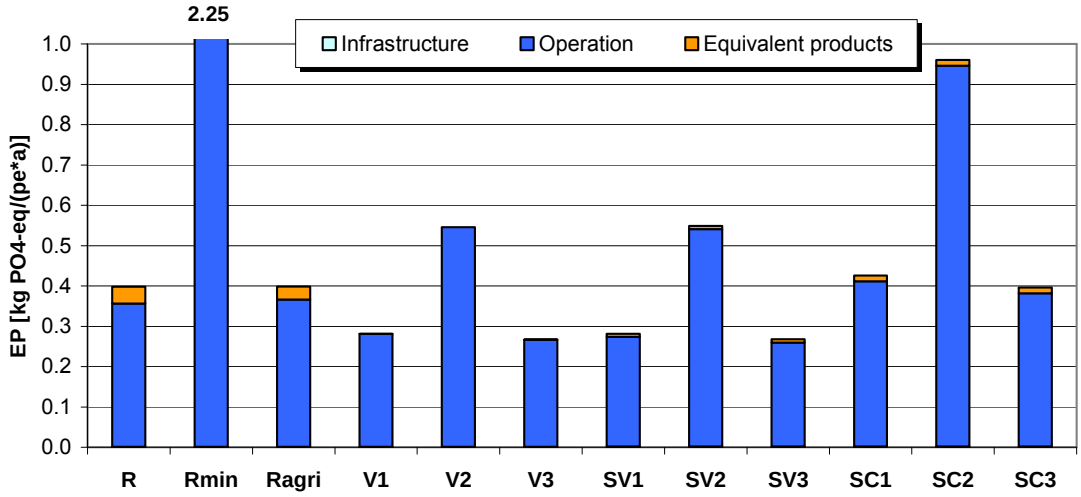

Figure 54: Eutrophication potential

The reference scenario with minimum standards for wastewater treatment (R_{min}) has an EP of 2.15 kg PO_4-equivalents per person and year. Per definition, this scenario represents conventional wastewater treatment without extended nutrient removal and consequently has the highest EP of all scenarios. It should be pointed out that this scenario still complies with all German legal regulations for wastewater treatment plants of this dimension. If extended nutrient removal is implemented in conventional wastewater treatment (scenarios R + R_{agri}), EP is reduced substantially (-81%) to 0.4 kg PO_4-eq/(pe*a). The scenarios with extended nutrient removal represent the capabilities of modern wastewater treatment technology. Despite the less stringent legal requirements, new or upgraded plants of the present dimension (5000 inhabitant equivalents) are often operated with denitrification and extended P elimination to minimize nutrient loads in the effluent.

For the separation scenarios, two aspects are decisive for eutrophication: a) the process for greywater treatment (SBR/MBR or soil filter) and b) the co-treatment of faeces filtrate in composting scenarios.

In composting scenarios, faeces are separated from flush water to obtain high dry matter content for the composting process. The flush water ("faeces filtrate") is heavily loaded with nutrients, particularly due to the high amounts of misled urine (30% of total urine) which is not separated properly in the toilets. This filtrate is treated together with greywater and substantially increases nutrient loads to the greywater treatment process. Composting scenarios with technical greywater treatment in SBR or MBR (SC1 + 3) do not offer advantages in EP compared to advanced wastewater treatment (R). The combination of faeces composting and GW treatment in soil filter (scenario SC2) even leads to a significant increase of EP (+140%), because the soil filter has only limited capacity for nutrient removal.

In vacuum scenarios, the complete mixture of faeces and flush water is treated together in the digestion process. In case of urine separation, the digester residual is dewatered

prior to stabilisation in open composting. The resulting sludge liquor is heavily loaded with nutrients (comparable to faeces filtrate). However, it is treated in a separate process (SBR with denitrification and chemical P elimination) close to the digester tank and not together with greywater. Hence, the nutrient load to greywater treatment is relatively low and results in better effluent quality than in composting scenarios: Scenarios with GW treatment in SBR/MBR (SV1 + 3) lead to a reduction of 30% in EP compared to the reference system R. Despite the low nutrient load, GW treatment in soil filter (SV2) again results in a significant increase in EP (+38% compared to R).

Vacuum scenarios without urine separation (V1-3) operate without any effluents from faeces treatment: the complete residual of the digestion process is directly applied in agriculture. Consequently, nutrient loads in the effluent are lowest for these scenarios (cf. chapter 5.1.4), but this benefit is partially offset by higher emissions of atmospheric NH_3. Hence, EP reduction potential between vacuum scenarios with and without urine separation is comparable (Figure 54).

The detailed contribution analysis of EP shows that for the soil filter scenarios, limited phosphorus removal (50% elimination) is the main reason for the increase in EP (Figure 55). In all other scenarios, the contribution of nitrogen to EP is larger than that of phosphorus, while COD generally plays only a minor role (< 10%). Atmospheric emissions of nitrogen gases are responsible for 20-30% of eutrophication except in those scenarios with lowest nutrient loads in the effluent: in scenarios V1 and V3, nitrogen gases contribute more than 50% to EP.

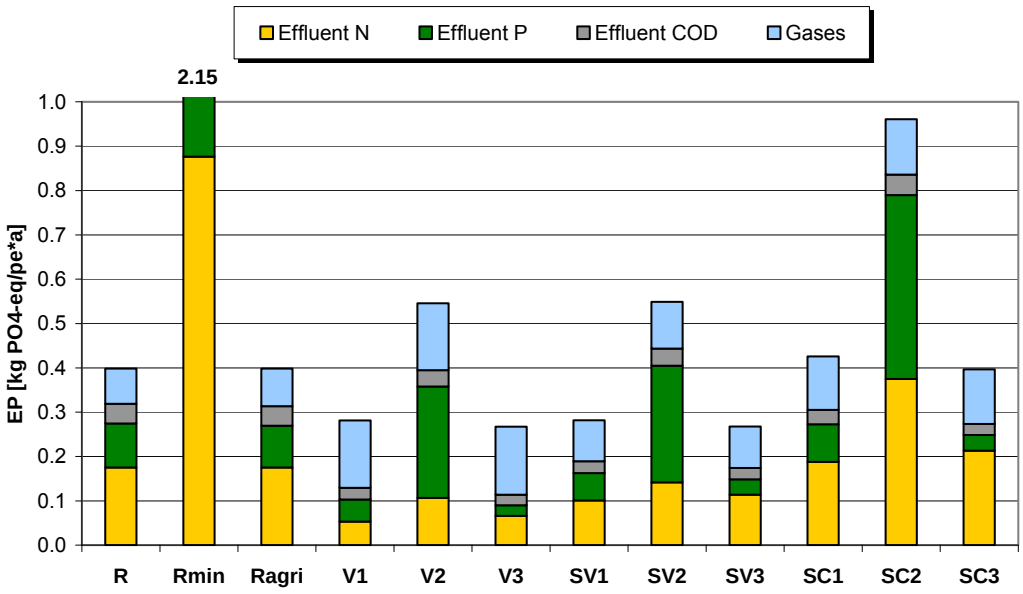

Figure 55: Contribution of effluent emissions of nitrogen, phosphorus and chemical oxygen demand and gaseous nitrogen emissions to eutrophication potential

Overall, the following conclusions can be drawn for the impact category of eutrophication:

- Composting systems do not decrease eutrophication potential due to the co-treatment of faeces filtrate (high volume = 24 L/(pe*d), high nutrient loads) with greywater.
- Soil filters are inferior to advanced wastewater treatment in nutrient removal, even if they treat only low-polluted greywater. The limited long-term retention of phosphorus is the main reason for this drawback.
- Vacuum systems using greywater treatment in technical systems with nutrient removal (SBR/MBR) offer benefits in EP. If digester residual is dewatered, sludge liquor (low volume = 6 L/(pe*d), high nutrient loads) should be treated in a separate technical process with high nutrient removal capacity and should not be mixed with greywater.
- The contribution of nitrogen gases to eutrophication is relatively small, but can become important for scenarios with low effluent loads and high NH_3 emissions (V1 + 3).
- All separation scenarios have a significantly lower eutrophication potential than conventional wastewater treatment without extended nutrient removal (R_{min}).

Thus, some separation systems can reduce the emission of nitrogen and phosphorus, while others may potentially increase them. Crucial points are a general load reduction to wastewater treatment due to the separate treatment of nutrient-rich toilet wastewater, the limited phosphorus elimination capacity in soil filters and the treatment options for concentrates (faeces filtrate, sludge liquor).

In general, the assumed elimination ratios for the different treatment processes have a strong influence on the results in this impact category (cf. chapter 5.1.4). Elimination ratios are qualified estimates for mean load-based elimination ratios based on data from pilot plants and literature. In sensitivity analysis, another approach to this issue is investigated: it is assumed that all scenarios with comparable technology (e.g. SBR) deliver the same effluent quality. Furthermore, two other indicators for this impact category are calculated based on site-dependent characterization factors of EDIP (Hauschild and Potting, 2003).

5.2.6 Human toxicity

This impact category evaluates all emissions which can have a toxic impact on human health, including emissions to air, water, and agricultural soil. The related indicator of human toxicity potential (HTP) ranges between 10 and 28 kg DCB-equivalents per person and year for all scenarios (Figure 56). In general, all three sub-parts of the systems (infrastructure, operation, and supply of equivalent products) contribute substantially to this indicator. Separation systems have significant benefits in HTP, decreasing total HTP by 53-63% with vacuum systems and 30-38% with composting systems compared to scenario R.

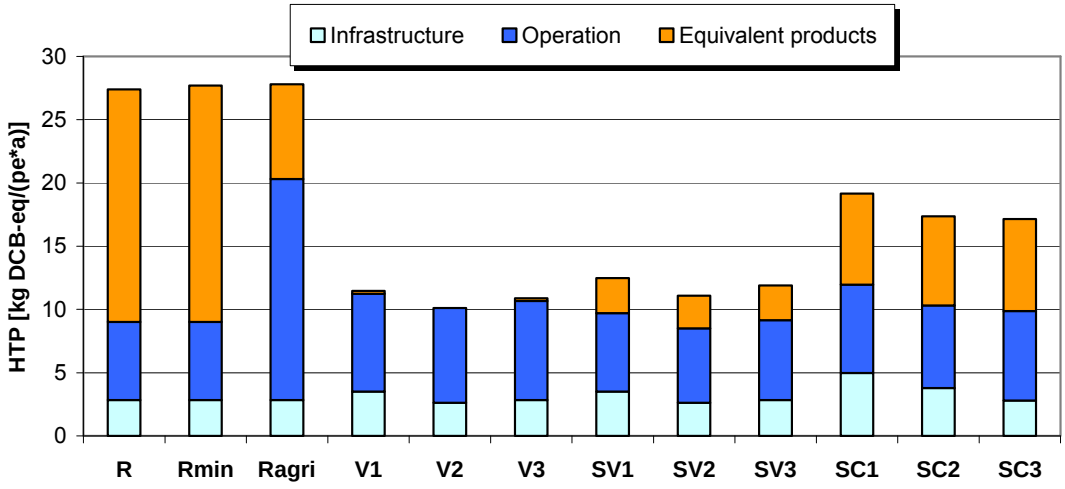

Figure 56: Human toxicity potential

A detailed contribution analysis reveals that the production and application of mineral fertilizer or sewage sludge causes a substantial part of HTP in the reference scenarios (Figure 57). In the reference systems without sewage sludge application in agriculture ($R + R_{min}$), the production and application of mineral fertilizer constitutes 62% of the total HTP, mainly due to the transfer of toxic heavy metals in mineral P fertilizer to agricultural soil. Additionally, the production of mineral P fertilizer causes considerable aquatic emissions of metals and especially fluoride from the processing of raw phosphate ores. If sewage sludge is applied in agriculture (R_{agri}), heavy metal content of the sludge contributes with 56% to the total HTP.

Interestingly, the total HTP of reference systems with or without sewage sludge application in agriculture is comparable. Thus, mineral fertilizer and sewage sludge are characterized by the same human toxicity potential, even though the specific heavy metal content of mineral fertilizer is much lower than that of sewage sludge (cf chapter 5.1.5). Process emissions during the production of mineral P fertilizer equalize these benefits of mineral fertilizer and result in an overall comparable HTP for all reference

scenarios. However, it has to be noted that process emissions of P fertilizer production usually occur at the place of phosphate mining (e.g. Morocco), whereas the pollution of agricultural soil takes place "on-site", i.e. in the proximity of the settlement. Therefore, the usage of mineral fertilizer shifts a part of the associated environmental burden to the country of fertilizer production.

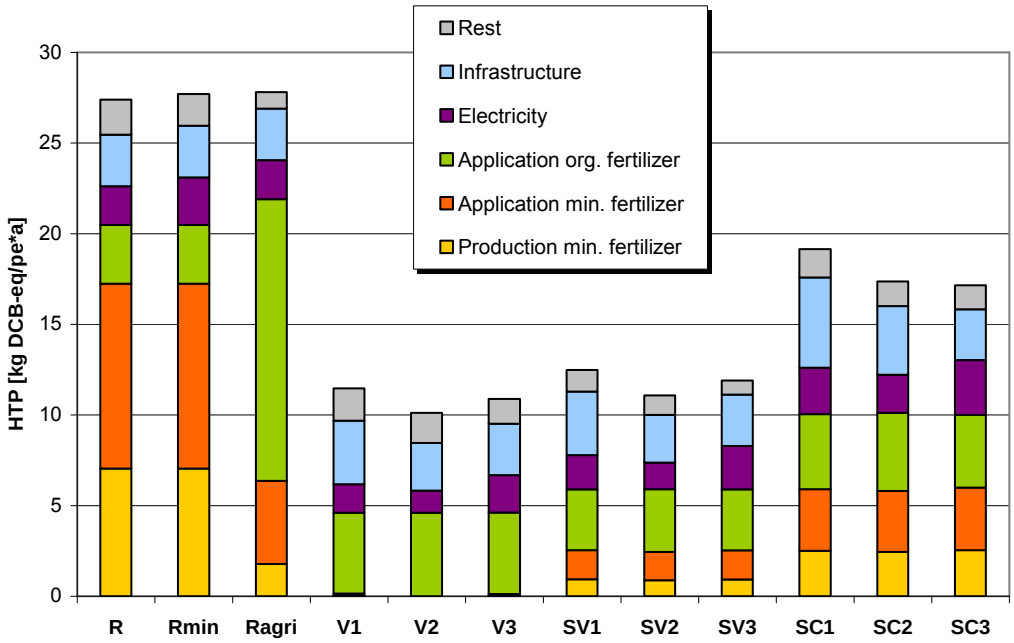

Figure 57: Contribution of sub-processes to human toxicity potential

Comparing conventional and separation systems, the substitution of mineral fertilizer or sewage sludge with organic fertilizers from faeces and urine leads to significant benefits in HTP for the separation systems. The transfer of toxic heavy metals to agricultural soil can be substantially reduced by applying secondary fertilizers with low heavy metal content. Additionally, toxic emissions from the production of mineral P fertilizer can be avoided. The minor increases in HTP due to the additional infrastructure do not offset the benefits of separation systems (Figure 57). Vacuum systems are superior to composting systems in HTP, because the substituted amount of mineral fertilizer is lower in the composting scenarios.

5.2.7 Freshwater ecotoxicity

For the impact category of aquatic ecotoxicity, emissions of eco-toxic substances to each environmental compartment (air, water, and soil) are relevant due to the inter-media transport of substances into surface waters. In this study, the respective indicator is mainly determined by the emissions of heavy metals, particularly by direct emissions into the water (from wastewater treatment) and indirect emissions via agricultural soil (from fertilizer application).

The calculated freshwater aquatic ecotoxicity potential (FAETP) of all scenarios is between 4 and 5.5 kg DCB-equivalents per person and year (Figure 58), i.e. in a relatively close range. An exemption is scenario R_{agri} which has an FAETP of 11 kg DCB-eq/(pe*a). In general, the major part of FAETP is caused by the operation of sanitation systems, with a small contribution from the supply of equivalent products (fertilizer).

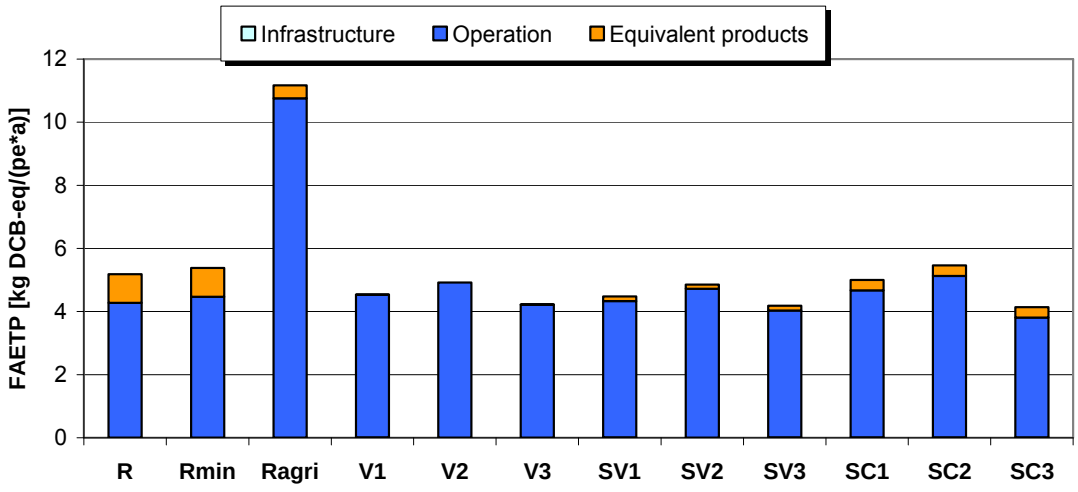

Figure 58: Freshwater aquatic ecotoxicity potential

The contribution analysis shows that direct emissions with the effluent of water treatment processes are responsible for the major part (52-66%) of FAETP in all scenarios, while the application of mineral and organic fertilizers contributes a smaller but relevant part to this indicator (29-43%). The significant increase in FAETP in scenario R_{agri} is due to sewage sludge application in agriculture (Figure 59).

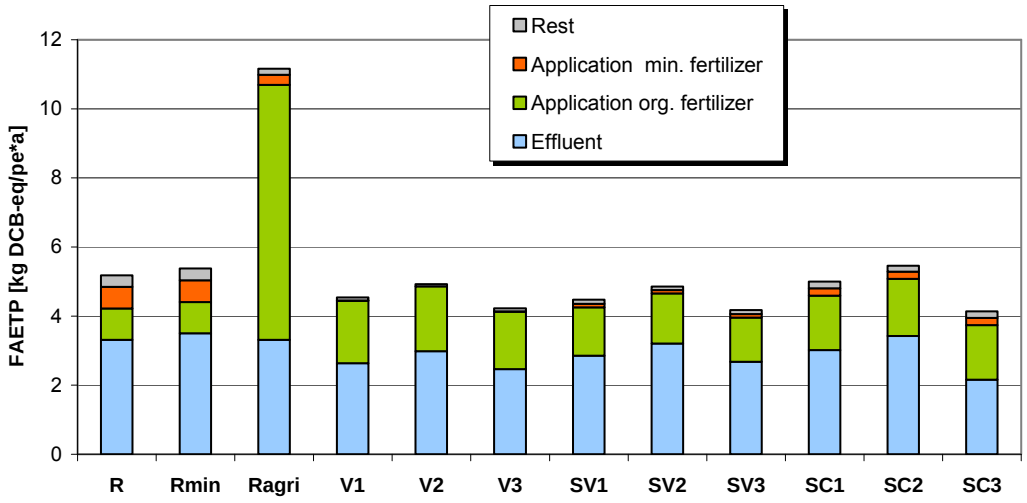

Figure 59: Contribution of sub-processes to freshwater aquatic ecotoxicity potential

The elimination of heavy metals in wastewater or greywater treatment is estimated with comparable elimination ratios for all treatment processes in the inventory. Thus, FAETP of direct heavy metal emissions to surface waters are roughly similar between all scenarios (Figure 59). MBR scenarios with greywater reuse (V3, SV3, SC3) have a slight advantage due to the partial substitution of drinking water with a high content of Cu and Zn. These two heavy metals have a distinct influence on this indicator, as both have high impact factors for aquatic ecotoxicity. Sewage sludge contains large amounts of Cu and Zn (particularly from input via drinking water, cf. Figure 42) and thus leads to the significant increase of FAETP in case of sewage sludge application in agriculture (R_{agri}).

The assumed concentrations of Cu and Zn in drinking water (0.16 and 0.37 mg/L, respectively) seem to have a strong impact on this indicator due to the transfer of these metals to agricultural soil via sewage sludge. For sensitivity analysis, the effect of a reduced concentration of Cu and Zn in drinking water is quantified for this impact category.

5.2.8 Terrestrial ecotoxicity

The second impact category for ecotoxicity evaluates toxic impacts on the terrestrial ecosystem. It is strongly determined by the transfer of heavy metals to agricultural soil with the different types of fertilizer. Emissions to air and water are also accounted for in the respective indicator, but they play a negligible role for the result in all scenarios (< 1-3% of total TETP).

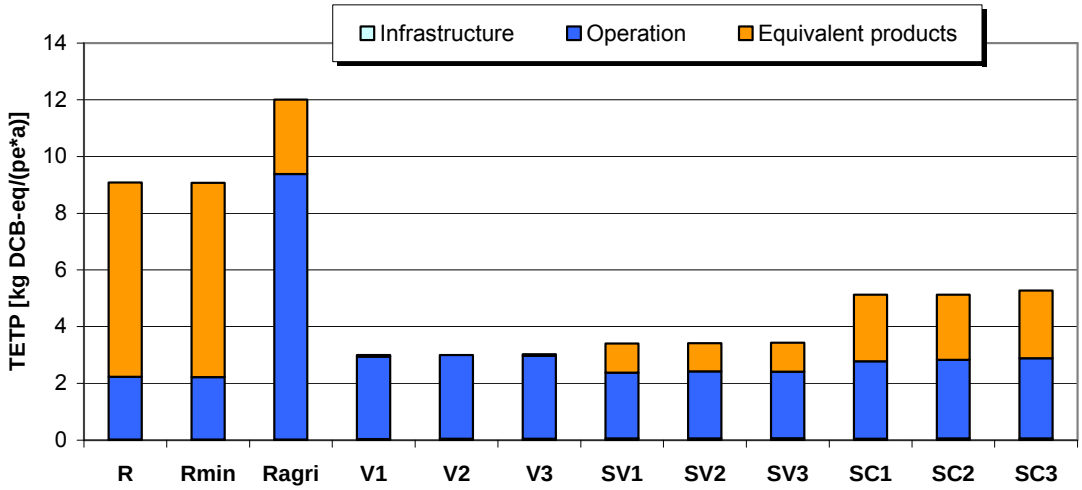

Figure 60: Terrestrial ecotoxicity potential

The comparison shows that terrestrial ecotoxicity is substantially reduced in separation scenarios (- 42-67%) by the substitution of mineral fertilizer with organic fertilizers from faeces and urine (Figure 60). Compared to the application of sewage sludge (scenario R_{agri}), the reduction potential for separation systems is even higher (- 56-75%). In other words, the heavy metal content of mineral fertilizer and sewage sludge causes a significantly higher terrestrial ecotoxicity potential than the organic fertilizers from faeces and urine.

This is confirmed by a contribution analysis for the different fertilizers to the indicator of TETP (Figure 61). In the reference systems, mineral fertilizer and sewage sludge are mainly responsible for TETP. For the separation systems, compost or digester sludge constitutes a major share of the total TETP. It has to be pointed out that separated urine has a negligible impact on TETP. Together with its high nutrient content (cf. chapter 5.1.2), the low heavy metal content of urine makes it a highly attractive fertilizer compared to other organic fertilizers such as faeces compost or digester sludge. From an ecotoxicological point of view, the separation of urine from faeces can be recommended.

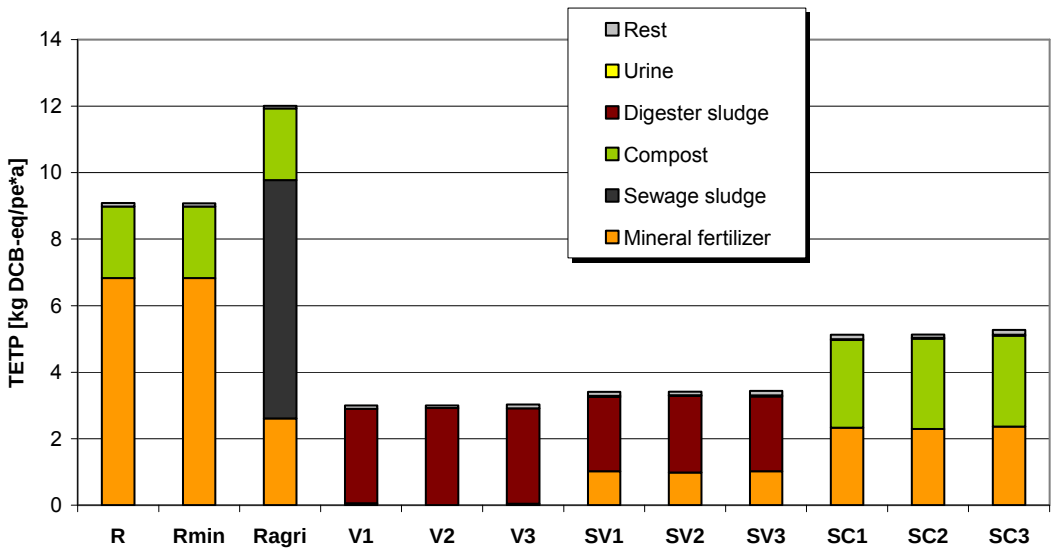

Figure 61: Contribution of different types of fertilizer to terrestrial ecotoxicity potential

5.2.9 Summary of LCIA results

This section summarizes all indicator results and relevant findings from Life Cycle Impact Assessment, thus giving an overview of the calculated results and the contribution analysis at a glance. Indicator results and findings are presented in different ways:

- Comparison of LCIA results for all indicators in relation to the reference scenario R (Figure 62)
- Comparison of LCIA results for all scenarios in relation to the reference scenarios R (Figure 63)

For a detailed analysis of each indicator results, the relevant chapters for the specific indicators can be referred to.

Comparison of LCIA results for all indicators

The overview of all LCIA indicators shows that the comparison between conventional and separation systems yields different profiles for each indicator (Figure 62):

- Resource-oriented indicators (CED, ADP) are ambiguous: most separation systems have benefits, but some scenarios have drawbacks.
- Emission-related indicators can be divided into indicators of macro-elemental emissions (C, N, P, S emissions for GWP, AP, EP) and those determined by micro-elements (heavy metals for HTP, FAETP, TETP).
- For macro-elemental emissions, the comparison between the conventional and separation systems is not conclusive: some indicators have benefits, while others show considerable drawbacks. The potential for acidification is significantly increased in all separation systems, while global warming can be reduced by the majority of them. Eutrophication is a twofold case: some scenarios decrease this impact considerably, while others will potentially increase it.
- Concerning micro-elemental emissions, all separation systems are superior in toxicity indicators due to the avoided heavy metal emissions to the environment.

This overview of all indicators gives a first hint on the difficulties in summarizing the results of this LCIA into one conclusive statement for or against separation systems. It seems to depend heavily on the respective impact category if a conclusive statement is possible or not.

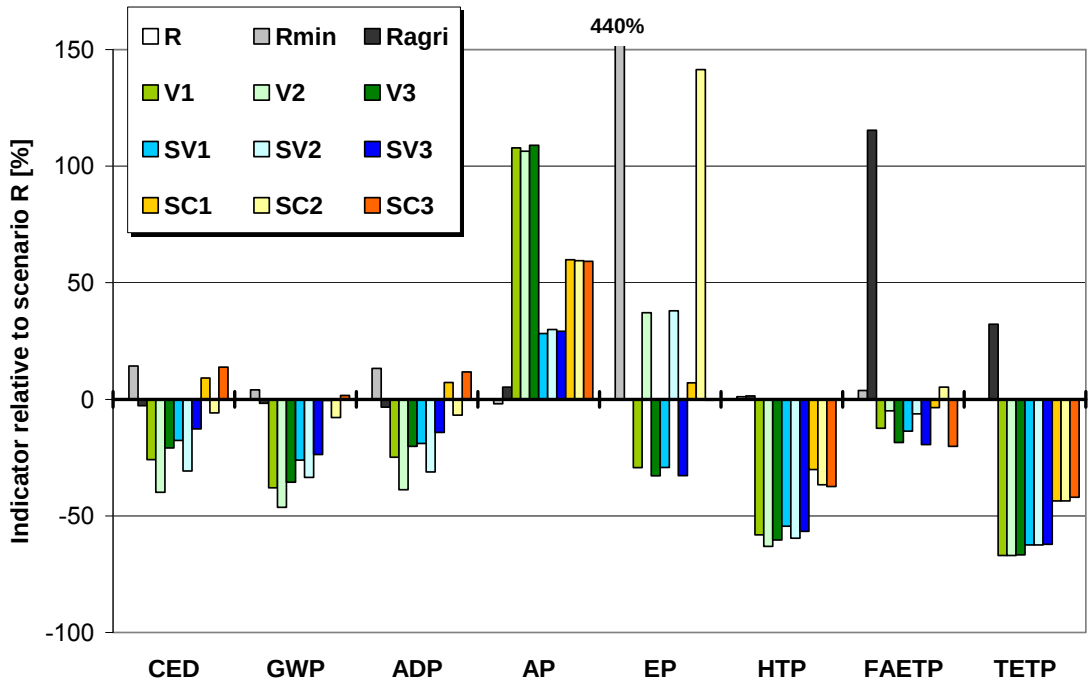

Figure 62: Comparison of all LCIA indicators (relative to scenario R)

Comparison of LCIA results for all scenarios

The comparison of all scenarios in relation to scenario R shows that some scenarios have benefits in the majority of impact categories, while others show a more distinct picture of benefits and drawbacks (Figure 63):

- Following its definition, the reference system with minimum standards (R_{min}) is clearly inferior to advanced wastewater treatment, especially in nutrient emissions, energy demand and related emissions.
- The agricultural application of sewage sludge (R_{agri}) leads to an increased ecotoxicity due to the transfer of wastewater-derived heavy metals to agricultural soil.
- Vacuum systems without urine separation (two-flow systems) show benefits in all impact categories except acidification. Only the soil filter scenario has a higher impact in eutrophication.
- Vacuum systems with urine separation (three-flow systems) also have benefits in most impact categories. The drawback of increasing acidification is less distinct than in the two-flow vacuum scenarios. Again, the soil filter leads to enhanced eutrophication.
- Composting systems are those separation systems with the smallest benefits. Benefits in toxicity indicators are smaller due to the lower amount of substituted mineral fertilizer. Energy demand is higher due to the unexploited energy recovery potential, so that only the soil filter scenario yields energetic benefits. The latter scenario leads to a significant increase in eutrophication due to high nutrient emissions in the effluent.

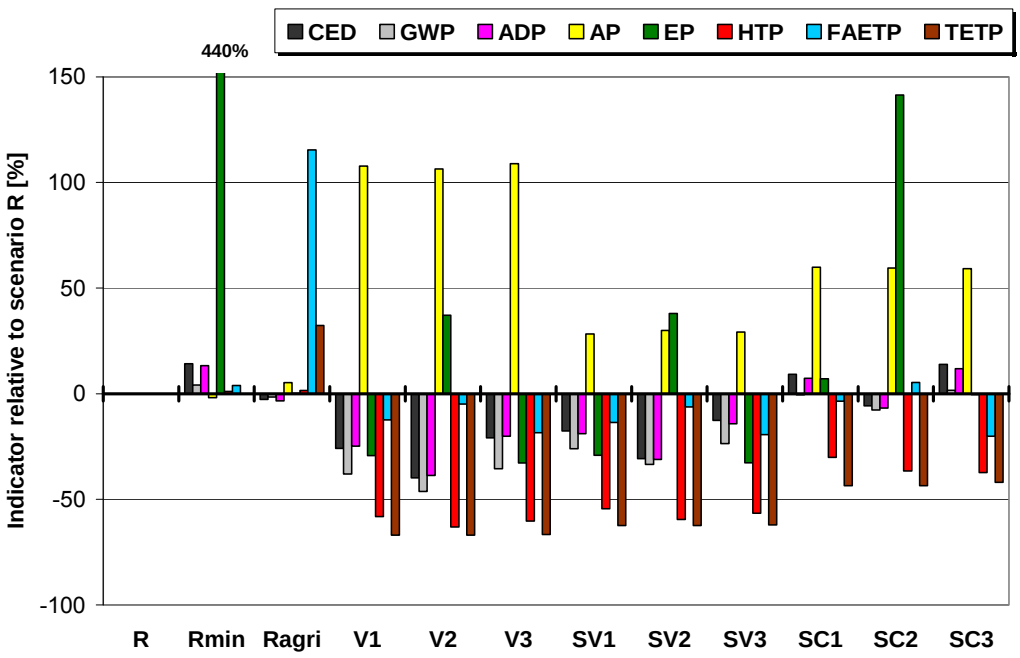

Figure 63: Comparison of all scenarios (relative to scenario R)

5.2.10 Normalization

In normalization, LCIA indicator results are related to the total impacts of the respective impact category in Germany. Thus, the relative share of each indicator result to the total actual environmental impact can be calculated. In fact, the normalized score of 1 pe*a inhabitant equivalent is equal to 100% of the total environmental impact, whereas 0.01 pe*a inhabitant equivalent means 1% of the total impact.

Normalization is an optional tool in valuation of LCIA to allow a first assessment of the quantitative importance of a certain indicator result in relation to the environmental situation in a society as a whole. It is stressed here that normalized indicator scores provide no final information on the importance of a certain indicator for the overall comparison.

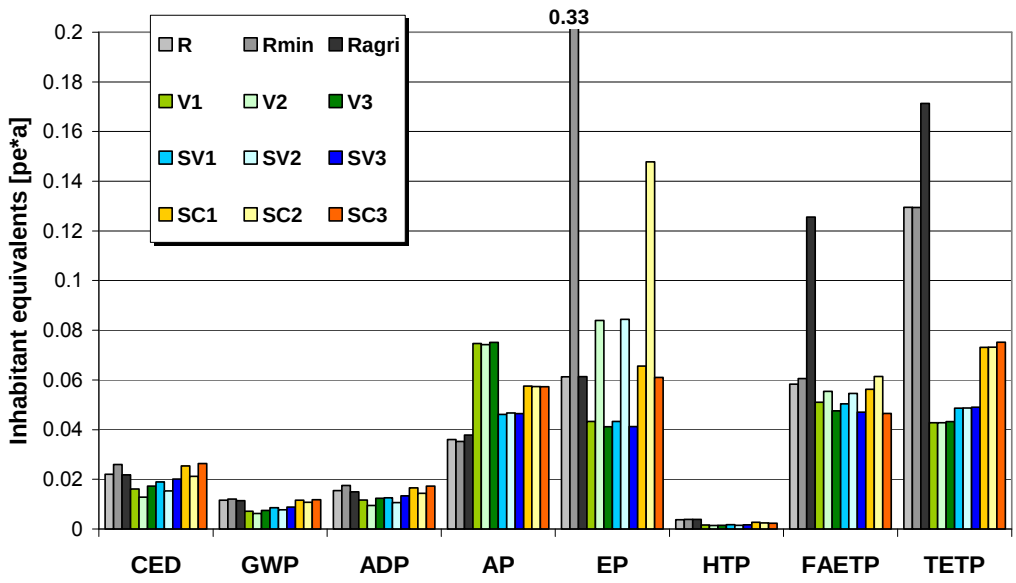

Figure 64: Normalization of all indicators from Life Cycle Impact Assessment

In detail, the normalized indicator scores (Figure 64) are evaluated as follows:

- Resource-related indicators (CED, ADP) have a low normalised score, accounting for < 3% of the total resource demand in Germany. Hence, wastewater management has only a marginal share of the total energy demand in Germany.
- The contribution to global warming is even lower: GWP normalized scores are below 1.2% for all scenarios. Considering the possible correlation of GWP to energy demand (CED), it seems that wastewater management has a relatively low GWP compared to its demand of non-renewable energetic resources.

- In acidification, the calculated normalized scores are relatively high (3.5-7.5% of total AP). This contribution is not caused by wastewater treatment itself, but mostly by application of different fertilizers and related emissions of NH_3. Agriculture is known to contribute a significant share to total NH_3 emissions in Germany (ECETOC, 1994).
- As expected, normalized scores for eutrophication are reasonably high, i.e. between 4-15% for all scenarios except the reference scenario with minimum standards (R_{min}). The latter scenario contributes with 33% to total eutrophication in Germany, mostly due to insufficient elimination of nutrients. If extended nutrient removal is not applied, municipal wastewater treatment contributes significantly to eutrophication. With extended nutrient removal, the share of municipal wastewater treatment can be minimized to ~ 6%. Diffuse pollution with nutrients is the major cause for eutrophication of surface waters in Germany (UBA, 2007b).
- Human toxicity has the lowest normalized score of all indicators (< 0.4%), indicating that wastewater management including the application of secondary fertilizers has a marginal toxic effect on humans compared with other activities in society. Thus, the effective protection of human health with common or future wastewater management options seems to be secured.
- For aquatic ecotoxicity, the normalized scores account to 4-6% of total FAETP for most scenarios. Loads of toxic heavy metals in WWTP effluent are mainly responsible for the ecotoxicological impact of wastewater management. If sewage sludge is applied in agriculture (R_{min}), the relative contribution rises significantly to 13% due to the possible transfer of wastewater-derived heavy metals to surface waters via agricultural soil.
- Normalized scores for terrestrial ecotoxicity are between 13-17% for the reference systems and 4-7% for separation systems. Input of heavy metals via mineral fertilizer or sewage sludge is mainly responsible for the high contribution of this impact category in the reference scenarios. Wastewater treatment itself has a marginal share of TETP, but the secondary functions of fertilizer supply extend the impacts of wastewater management to agriculture, thus causing a significant impact on terrestrial ecotoxicity.

In summary, normalization of indicator results showed a marginal share of wastewater management for energy and resource demand, global warming, and human toxicity. Moderate to high contribution to total environmental impacts is calculated for acidification, eutrophication, and ecotoxicity.

5.2.11 Grouping and Weighting

Grouping and weighting of indicator results from LCIA is another optional step of valuation. It can be helpful to come to a conclusive statement while comparing two scenarios in their environmental impacts. Here, a modified evaluation method based on the UBA method (Schmitz and Paulini, 1999) is applied, adding toxicity indicators to the original method. It is based on the ranking of the various impact categories considering ecological hazard, distance to target, and specific contribution of each indicator.

This method is used for a direct comparison of two scenarios. Out of the large number of possible combinations, three combinations are chosen to exemplify the valuation procedure and possible outcomes in detail.

Comparison of scenario R vs V1 (FEASIBLE)
Scenario V1 is a separation system with a relatively low level of complexity: toilet wastewater is collected by a vacuum system, digested and applied to agriculture, while the remaining greywater is treated in a conventional activated sludge plant. The actual implementation of such a system is not very different in infrastructural needs to a conventional system: it just requires a second pipe network and vacuum toilets. Furthermore, all required process technology (vacuum drainage, biogas plant, SBR) is well-known and thus easy to handle. Hence, the feasibility of scenario V1 is estimated to be high compared to other separation scenarios.

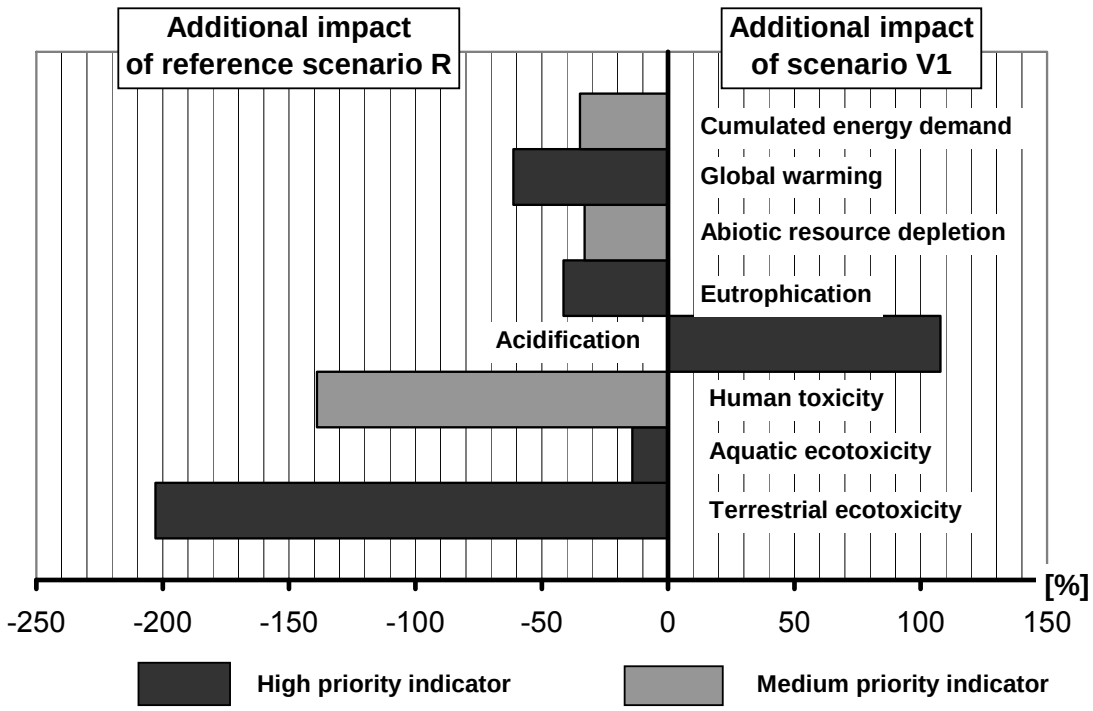

Figure 65: Comparison of scenario R and V1 with valuated indicator results in T diagram

The direct comparison between scenarios R and V1 shows distinct benefits for the separation scenario after valuation (Figure 65). Except for acidification, environmental impacts of the reference scenario are all significantly higher. If indicators with similar priority are offset against each other, the valuated comparison leads to a conclusive result: scenario V1 is clearly favourable in terms of environmental impacts.

Comparison of scenario R vs SV3 (SOPHISTICATED)

In contrast, scenario SV3 can be described as the most sophisticated separation scenario in terms of technical effort. It employs vacuum drainage, urine separation, collection and treatment, and greywater reuse. Thus, four different pipe networks are required and a variety of technical processes of which some have not been tested successfully in larger scale (e.g. vacuum separation toilets, urine treatment, greywater reuse). The feasibility of a large-scale implementation of scenario SV3 is probably limited in the near future.

However, the direct comparison between scenarios R and SV3 also yields distinct benefits for the separation scenario (Figure 66). Offsetting the indicators with comparable priority, scenario SV3 is superior to the reference system: drawbacks in acidification are clearly outweighed by the benefits in energy and resource demand, global warming, eutrophication, and toxicity. Scenario SV3 is favourable in terms of environmental impacts compared to the reference system.

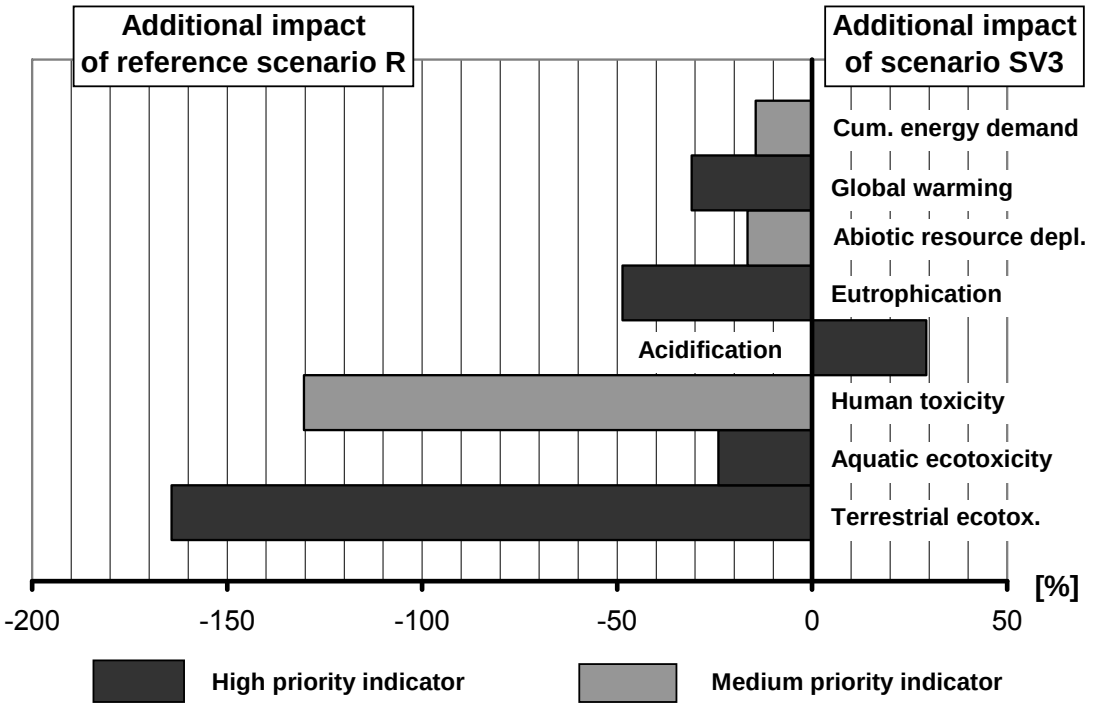

Figure 66: Comparison of scenario R and SV3 with valuated indicator results in T diagram

Comparison of scenario R vs SC2 (LOW-TECH)

Scenario SC2 is a separation system using low-tech natural processes. Faeces are drained by gravity and processed in composting after dewatering, whereas greywater is treated in a soil filter. Only the treatment of separately collected urine may pose some technical difficulties in this scenario.

Here, the valuated comparison of indicators does not result in a distinct statement of environmental benefits for this separation system (Figure 67). Drawbacks in acidification and eutrophication are critical and do not offset benefits in indicators with the same priority (GWP, TETP). Even though the separation scenario has benefits in all indicators with medium priority, the overall comparison cannot come to a conclusive statement, following the definition of UBA (Schmitz and Paulini, 1999). In contrast, the result of this comparison has to be declared as inconclusive, because the environmental benefits of medium priority indicators (reduced resource demand and human toxicity) cannot be compensated directly with the drawbacks in high priority indicators.

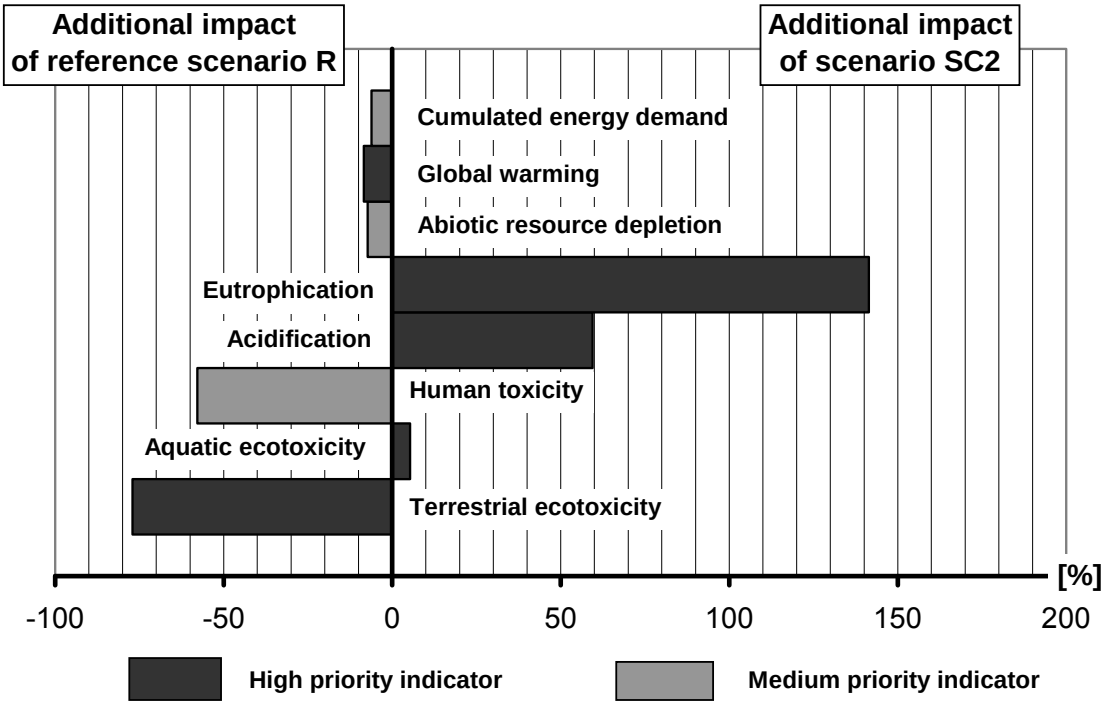

Figure 67: Comparison of scenario R and SC2 with valuated indicator results in T diagram

Remaining separation scenarios

For completion, all separation scenarios are compared to the reference scenario, using the valuation procedure described above. The respective results in the form of T diagrams with valuated indicator results are presented in annex 12.10. The extent of this study does not allow a verbal comment for each comparison, but the graphical results are attached for information of the interested reader.

5.3 Sensitivity analysis

In sensitivity analysis, the robustness of the results is tested by varying certain parameters of the Life Cycle Inventory and calculating their effect on the indicator results of LCIA. Due to the high number of parameters in the inventory, the sensitivity analysis is limited to those parameters which have been identified during contribution analysis of LCIA indicators, presumably having a distinct effect on the results. Table 64 presents an overview of the tested parameters and the respective indicators. In addition to the sensitivity analysis for inventory data, four additional impact indicators and an alternative valuation method are calculated to reveal the influence of indicator choice and LCIA methodology on the outcomes of this LCA.

Table 64: Parameters and indicators for sensitivity analysis

Observed Indicators	CED	GWP	EP	AP	FAETP	TETP
Data of Life Cycle Inventory						
Energy recovery without biowaste	X	X				
Transport distance of organic fertilizers	X	X				
Energy demand of urine treatment	X					
Energy demand of vacuum plant	X					
Efficiency of urine separation toilets	X	X	X	X		
Volume of reused greywater and energy demand for drinking water	X					
Effluent concentrations of SBR			X			
NH_3 emissions in application of liquid organic fertilizer				X		
Cu/Zn content in drinking water					X	X
Heavy metal data of mineral fertilizer						X
Plant availability of P in sewage sludge						X
Alternative indicators						
Aquatic eutrophication (AEU)*			X			
Terrestrial eutrophication (TEU)*			X			
Aquatic ecotoxicity (AET)**					X	
Terrestrial ecotoxicity (TET)**						X
Valuation with original UBA method						

** EDIP 2003 (Hauschild and Potting, 2003)*
*** IMPACT 2002+ (Jolliet et al., 2003a)*

5.3.1 Energy recovery without biowaste

If biowaste is not co-digested together with toilet wastewater, the amount of energy that can be recovered from organic matter is significantly lower (-17 kWh/(pe*a)). For most energy recovery scenarios, total CED is higher than the reference scenario without biowaste co-digestion (Figure 68). Only those scenarios with low-energy greywater treatment (V2 and SV2) can maintain an energetic benefit without biowaste co-digestion. For global warming, benefits of energy recovery are significantly reduced without biowaste, but energy recovery scenarios are still superior to the reference system (Figure 69).

These calculations underline the importance of an integrated approach treating both toilet wastewater and biowaste if energy recovery is targeted. Biowaste contains a high amount of organic matter (= energy recovery potential) and thus improves the overall energy balance of the separation scenarios decisively. Without biowaste co-digestion, the high energy demand for operating the vacuum plant and the digestion process offsets the recovered energy from toilet wastewater, leading to a higher total CED.

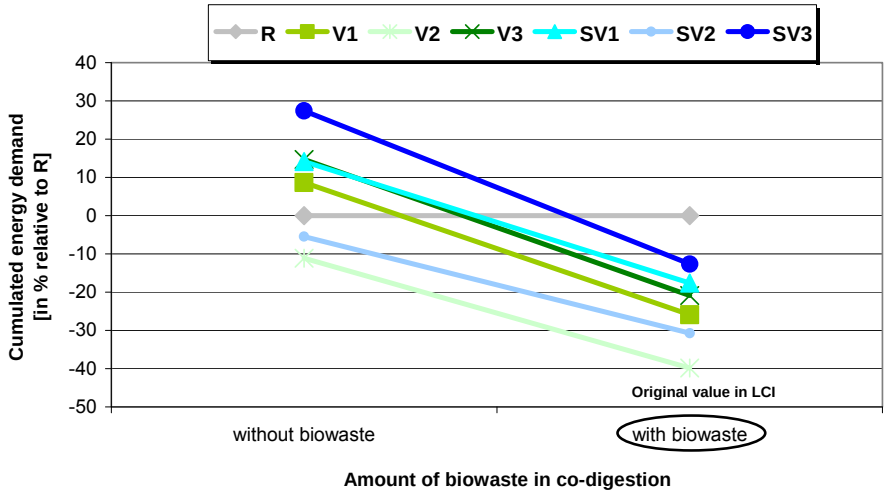

Figure 68: Variation of CED with and without biowaste in co-digestion

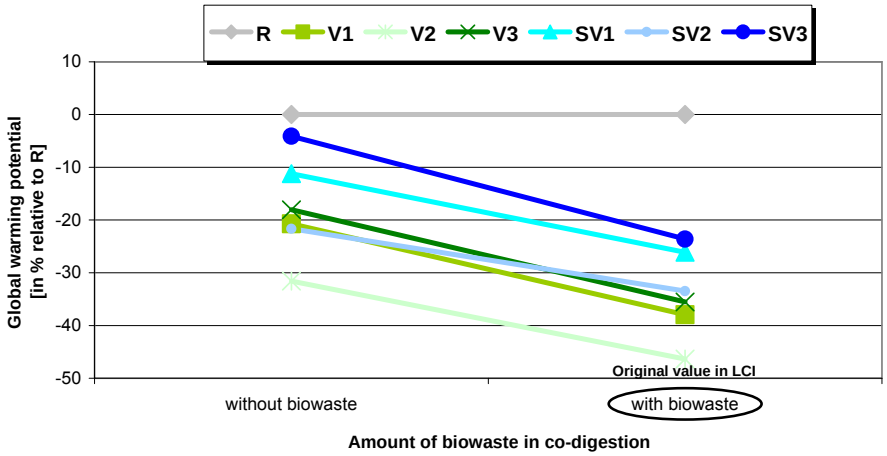

Figure 69: Variation of GWP with and without biowaste in co-digestion

5.3.2 Transport distance of organic fertilizers

Transport distance of organic fertilizers (urine, compost from faeces and biowaste, digester sludge, sewage sludge) is set to 20 km in the inventory, thus representing the application of secondary fertilizers in a relatively short distance to the settlement. If no suitable agricultural land is available in the proximity, this transport distance might be increased considerably.

If transport distance is increased to 100 km, cumulated energy demand and global warming potential rise due to the additional fuel required for the trucks. However, this effect is relatively small in the urine separation scenarios SV and SC: CED increases by 2% and GWP by 4%, respectively (Figure 70 and Figure 71). This increase does not lead to a different qualitative result of the comparison of urine separation scenarios with the reference scenario R. In vacuum scenarios without urine separation (V), the complete volume of urine, faeces, and flush water is transported to the fields after the digestion process. The effect of a longer transport distance is significant: the increase of CED and GWP amounts to 13 and 26%, respectively, for a transport distance of 100 km. Hence, the benefits of V scenarios in CED and GWP are partially offset if digester sludge is transported over long distances. The transport distance up to which the vacuum scenario V1 is still superior to the reference system are calculated to 189 km for CED and 156 km for GWP.

Overall, the transport distance has a significant influence in vacuum scenarios without urine separation (V), whereas its influence in urine separation scenarios SV and SC can be described as minor. The volume of urine and faeces itself in relation to their specific nutrient content justifies a transport over longer distances, even for urine with relatively high volume. If flush water is not separated from these fertilizers, transport should be limited to a minimum.

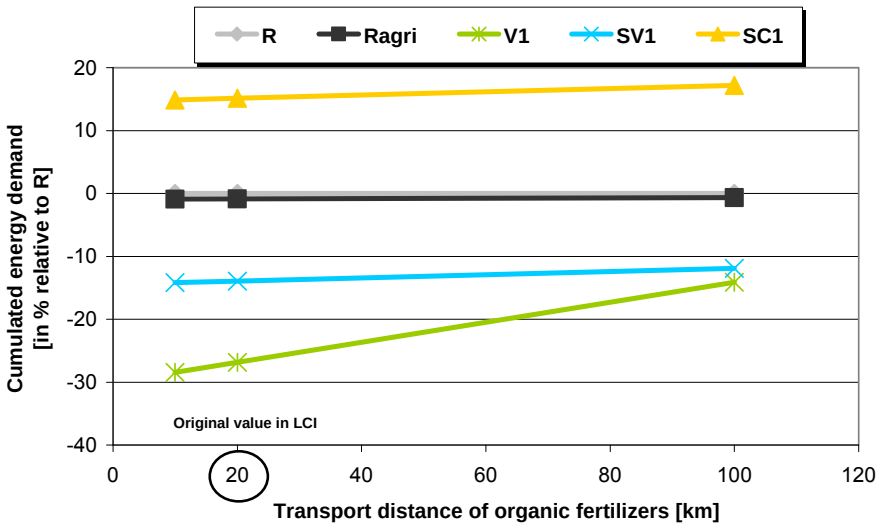

Figure 70: Variation of CED with transport distance of organic fertilizers

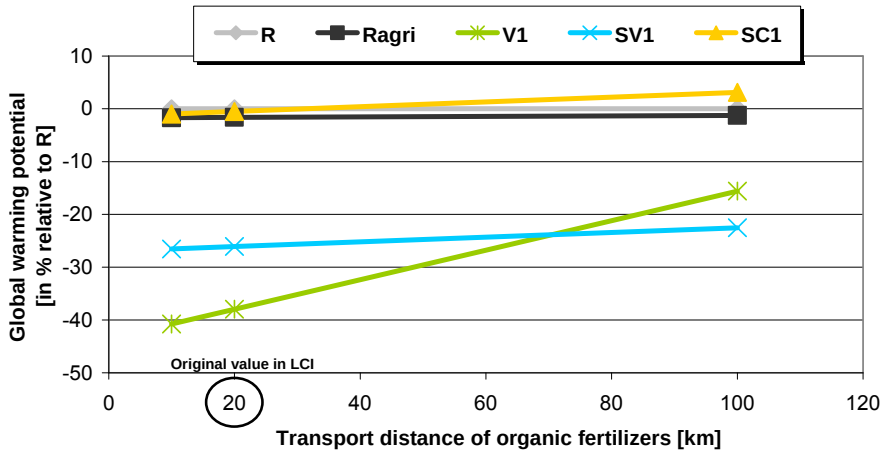

Figure 71: Variation of GWP with transport distance of organic fertilizers

5.3.3 Energy demand of urine treatment

The treatment of separated urine for the elimination of organic micropollutants is an important feature to allow its agricultural application without potential risks for humans or the agro-ecosystem. However, the treatment of urine has not been tested in full-scale, so that the energy demand had to be roughly estimated in the inventory (1 kg ozone per m³ = 15 kWh/m³).

If this energy demand is doubled to 30 kWh/m³, cumulated energy demand rises by 6% for all urine separation scenarios compared to the reference scenario R (Figure 72). A qualitative change in the comparison is detectable only for scenario SC2. If no urine treatment is applied (= 0 kWh/m³), the results of the qualitative comparison are stable for all scenarios.

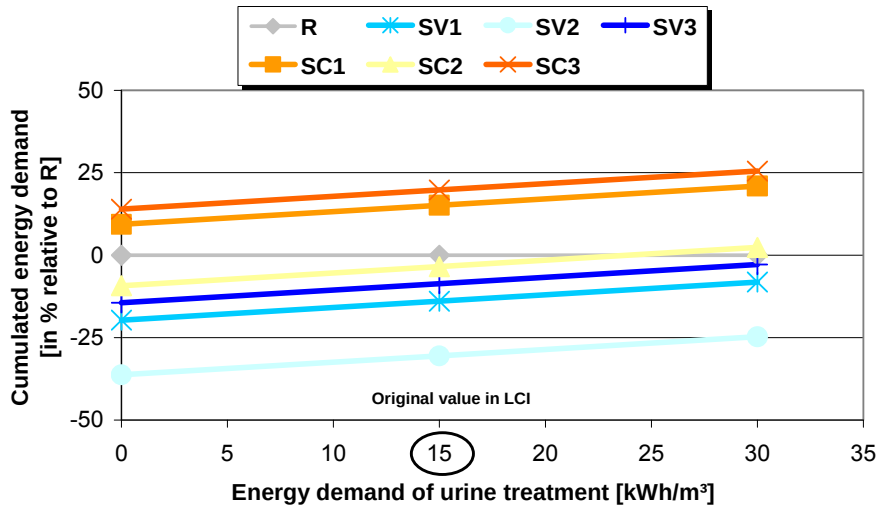

Figure 72: Variation of CED with energy demand of urine treatment

In all, the energy demand of urine treatment does not have a decisive influence on the energetic comparison if estimated in realistic ranges (15-30 kWh/m³). It has to be noted though that urine separation does not lead to energetic benefits if combined with faeces composting (SC1/3), even if no treatment is considered.

5.3.4 Energy demand for vacuum plant

The energy demand for the operation of the vacuum plant is estimated to 15 kWh per inhabitant and year in the LCI. However, literature data shows a wide range of possible values, with a minimum value of 3.1 kWh/(pe*a) postulated in the Berlin pilot project and a maximum of 51 kWh/(pe*a) for the pilot plant in Flintenbreite (cf chapter 4.1.2.3). Hence, the influence of a variation in energy demand for operating the vacuum plant is calculated for the indicator CED, covering a range of 7.5 to 30 kWh/(pe*a).

If the energy demand of the vacuum plant is increased to 30 kWh/(pe*a), CED of vacuum scenarios rises by 45% compared to the reference scenario R (Figure 73). Thus, all separation scenarios are now inferior in CED to the reference scenario. The respective maximum energy demand of the vacuum plant for realizing an energy benefit ranges from 18 to 28 kWh/(pe*a) for scenario SV3 and V2, respectively.

The vacuum plant is one of the major energy consuming processes in energy recovery scenarios, partially offsetting the energetic benefit of these systems in the comparison with the conventional system. Therefore, its energy demand should be minimized to recover a net maximum amount of energy from wastewater-derived organic matter. A reduction of the energy demand to 7.5 kWh/(pe*a) reduces CED of separation systems by 22% compared to the reference scenario R.

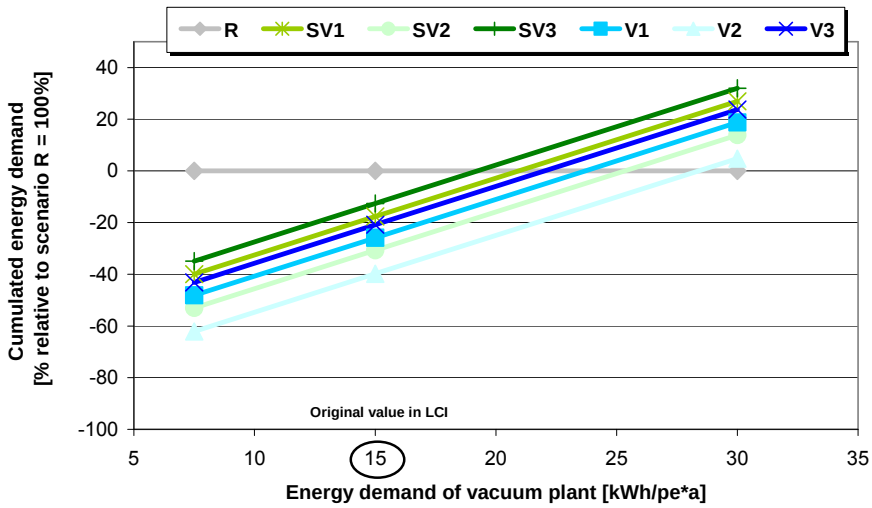

Figure 73: Variation of CED with energy demand of vacuum plant

5.3.5 Efficiency of urine separation toilets

The separation efficiency of the urine separation toilets is an important parameter in urine separation scenarios, determining the relative amount of undiluted urine that can be collected and applied as fertilizer in agriculture. However, the correct long-term operation of the separation toilets depends on technical issues (valve operation, blockages of urine pipes with precipitates etc) as well as user behaviour. Therefore, the estimation of average separation efficiency is affected with uncertainty. In this study, it is assumed that 70% of the daily urine can be effectively separated with separation toilets. However, the actual efficiency during long-term operation could be significantly higher (due to improved toilet design or user education) or lower (due to improper usage and maintenance of the separation mechanism).

The efficiency of urine separation has an effect on a variety of LCIA indicators: energy demand and global warming are affected through the amount of substituted fertilizer and the related energy demand. Acidification is affected by urine application causing considerable NH_3 emissions, whereas additional misdirected urine can lead to increased eutrophication by increasing nutrient loads of faeces filtrate which is eventually treated with greywater. All relevant changes in system expansion processes (amount of substituted fertilizer and energy) are considered during the calculation of this sensitivity analysis.

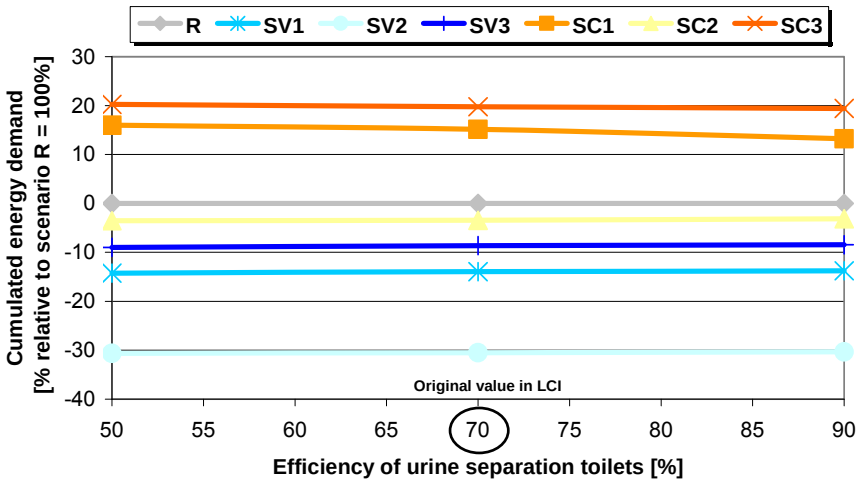

Figure 74: Variation of CED with efficiency of urine separation toilets

Variation of CED with urine separation efficiency is marginal for all scenarios (Figure 74): if the separation efficiency is increased to 90%, CED of urine separation scenarios slightly decreases for scenarios SC1 (-2%) and SC3 (-0.3%) due to a higher amount of substituted N fertilizer. In vacuum separation scenarios (SV), CED slightly increases with separation efficiency (+0.3%), probably due to the missing organic matter of urine in digestion. However, the qualitative comparison to reference scenario R does not change for any of the urine separation scenarios.

Changes in global warming are more significant: GWP decreases significantly with increasing separation efficiency. An improvement of separation efficiency to 90% leads to a 4-5% reduction of GWP, whereas a decline to 50% efficiency results in a 4-6% increase in GWP of separation scenarios. However, composting scenarios SC1 and SC3 change their relative comparison to the reference system from slightly worse (50% separation) to slightly better (90% separation).

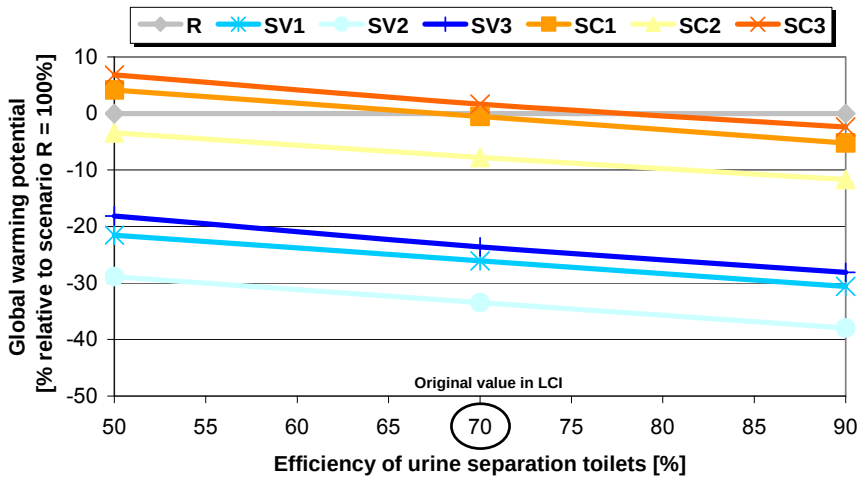

Figure 75: Variation of GWP with efficiency of urine separation toilets

The acidification potential gives a twofold picture (Figure 76): in vacuum separation scenarios, AP significantly decreases by 11% with better separation (90%), presumably due to less nitrogen emissions in open composting of digester sludge: properly separated urine does not contribute to NH_3 emissions from processing of the digester residual. This amount of avoided NH_3 emissions offsets an increase in NH_3 emissions during urine application. In contrast, scenarios SC1 and 2 show a small increase in AP (+3%) when increasing the separation efficiency to 90%. Here, the increased NH_3 emissions from urine application are responsible for the higher AP. Scenario SC3 has an optimum: higher and lower separation lead to an increased AP. In all, the separation efficiency has no influence on the qualitative comparison of urine separation scenarios with the reference system: all separation scenarios have a significantly higher impact in AP, independent of the separation efficiency.

In eutrophication, the increase of separation efficiency results in a reduction in EP for all scenarios (Figure 77). The reduction is more significant in composting scenarios SC1 and 3 (-29%) and especially for the soil filter scenario SC2 (-74%). The co-treatment of faeces filtrate (containing misdirected urine with high nutrient content) with greywater is responsible for this effect, particularly pronounced in soil filter scenario with low nutrient elimination. Nevertheless, scenario SC2 remains inferior in EP even in case of high separation efficiency (90%), whereas scenarios SC1 and 3 become superior to the

reference system in EP. For vacuum separation scenarios, the effect in EP is not that strong (-7% in EP fro 90% separation). Here, misdirected urine ends up in sludge liquor which is treated in a separate process with high nutrient elimination.

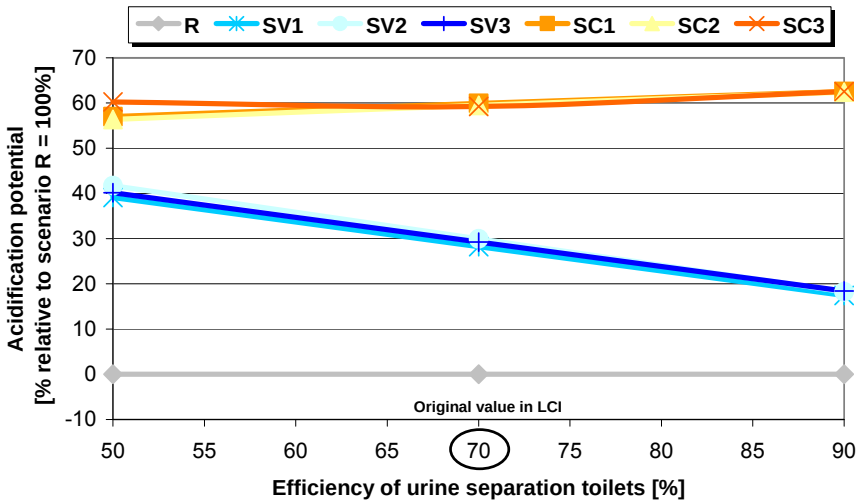

Figure 76: Variation of AP with efficiency of urine separation toilets

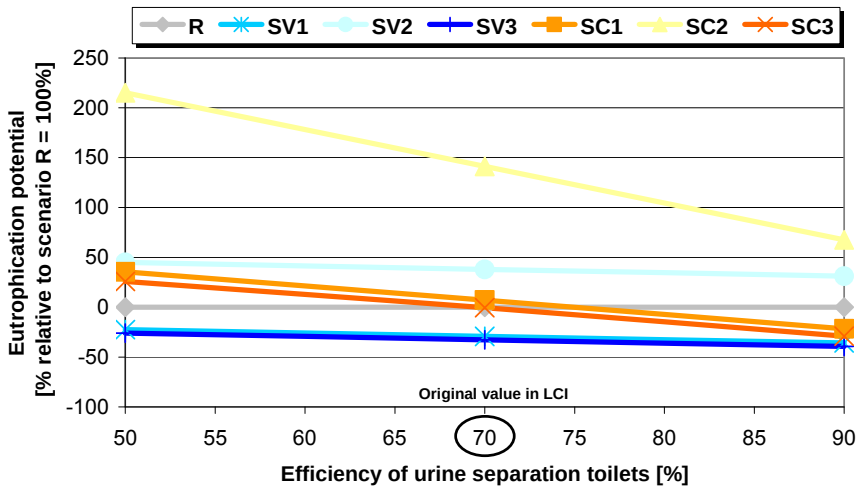

Figure 77: Variation of EP with efficiency of urine separation toilets

In all, an increase in urine separation efficiency can have a distinct effect on GWP, AP and EP, thus changing the results of the comparison for certain impact categories. The energetic comparison (CED) is not influenced much by this parameter. Nevertheless, the optimization of the separation process itself should be in the focus of technical development of urine separation.

5.3.6 Reuse volume of and energy demand for water supply

The simplified feasibility of non-potable reuse of purified greywater is an important feature of separation systems. However, it was found that the substitution of drinking water in toilet flushing does not lead to an energetic benefit in reuse scenarios V3, SV3, and SC3. The volume of substituted drinking water is too low with 5.4 L/(pe*d) using vacuum toilets and 24 L/(pe*d) using gravity toilets. Additionally, the energy demand for drinking water production is estimated relatively low (0.5 kWh/m³) in relation to the pumping energy for the reused greywater (0.27 kWh/m³). Thus, the variation of CED with the volume of reused greywater is calculated assuming simple (0.5 kWh/m³) or complex drinking water treatment (1 kWh/m³) to determine the conditions under which reuse of greywater can be beneficial in energy demand.

For simple drinking water treatment, a volume increase to 50 L/(pe*d) of substituted drinking water results in a 4% decrease of CED for vacuum and 2% for gravity systems (Figure 78). This does not result in a change in the relative comparison to the reference system or the separation systems without reuse. Assuming complex drinking water treatment, increased reuse volume results in a 9% increase in CED for vacuum and 5% for separation systems (Figure 79). Now, scenario SV3 and V3 are energetically favourable to the reference system and to the respective separation scenarios without reuse (SV1 and V1).

In all, it can be concluded that the reuse of purified greywater only results in energetic benefits if a complex drinking water treatment is assumed and the volume of substituted drinking water is high (> 25 L/(pe*d), i.e. more than 30% of total greywater flow). Otherwise, the additional energy demand during treatment (MBR) and the pumping energy for delivery of reused greywater back to the households offset the energetic benefits of drinking water substitution.

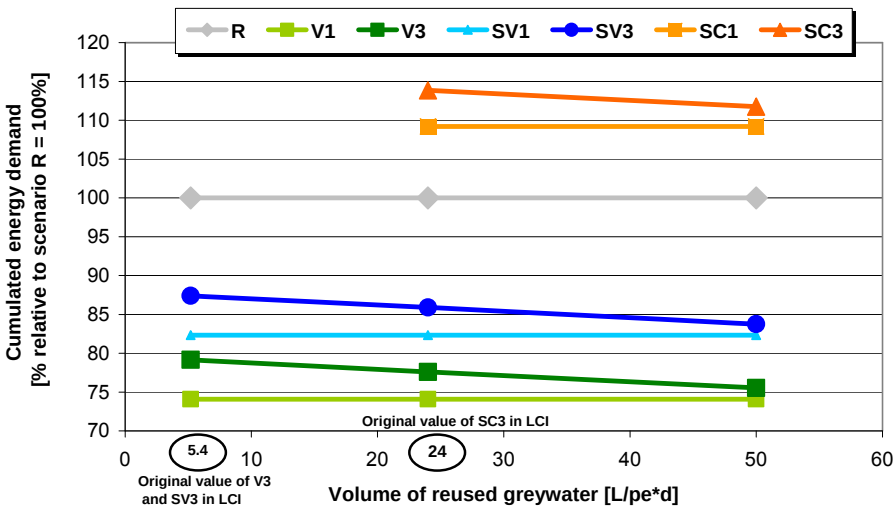

Figure 78: Variation of CED with volume of reused greywater

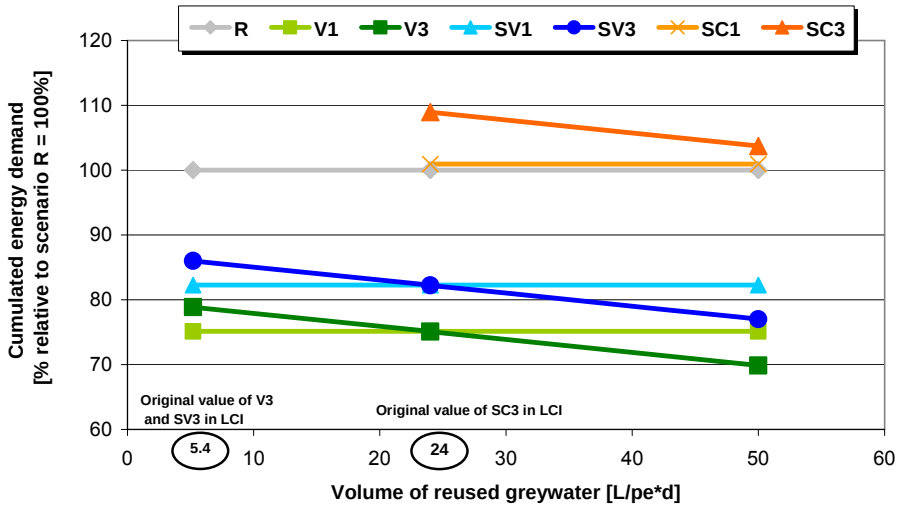

Figure 79: Variation of CED with volume of reused greywater, assuming complex drinking water treatment (1 kWh/m³)

5.3.7 Comparable effluent concentrations in SBR

Effluent concentrations of activated sludge plants in scenarios R, V, SV, and SC are different in conventional and separation scenarios, because elimination ratios for COD, N and P are estimated for each scenario separately based on total loads of influent to effluent. This refers to wastewater and greywater treatment plants (SBR or MBR), but also to the treatment of concentrates (sludge liquor of digestion). However, the actual elimination ratio of a technical process also depends on the operational parameters, e.g. hydraulic retention time, aeration, or applied amount of chemicals (Fe for precipitation). Consequently, another systematic approach to this issue would be to assume equal effluent concentrations for comparable technical processes (SBR and MBR) if operational parameters are adapted to influent loads.

Hence, it is assumed that all SBR processes (wastewater treatment in scenario R, greywater treatment in V1/SV1/SC1, and sludge liquor treatment in SV2 and SV3) have a comparable effluent quality of 50 mg/L COD, 1 mg/L NH_4-N, 7 mg/L NO_3-N, and 0.6 mg/L PO_4-P. Energy and chemical demand is adjusted accordingly. For MBR plants in reuse scenarios V3/SV3/SC3, effluent concentrations are assumed to 25 mg/L COD, 1 mg/L NH_4-N, 7 mg/L NO_3-N, and 0.3 mg/L PO_4-P. The low concentrations of COD and PO_4-P are justified by the superior particle retention capacity of membrane processes, so that retention of small sludge particles or Fe precipitates is better in MBR compared to SBR processes. For the soil filters, effluent concentrations are not adjusted, because their elimination performance can only be partially controlled by operational parameters.

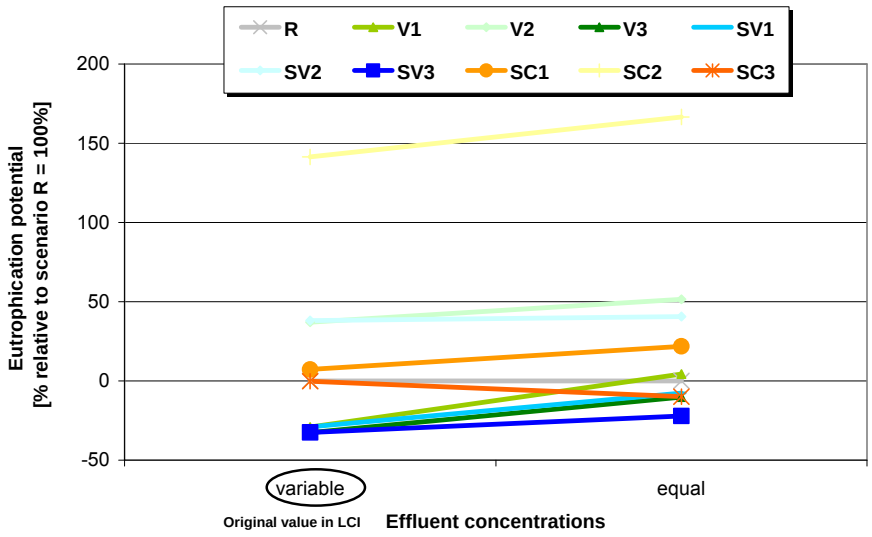

Figure 80: Variation of EP with definition of effluent concentrations of SBR and MBR

With equal effluent concentrations in SBR and MBR plants, the calculated eutrophication potential of separation systems is significantly different in relation to the reference scenario R (Figure 80). EP of most separation scenarios increases compared to scenario R, while only scenario SC3 has a lower EP while assuming equal concentrations. Scenario V1 is now slightly inferior to the reference system in EP, as well as all soil filter scenarios and scenario SC1. Benefits in EP are calculated for reuse scenarios V3, SV3, and SC3 due to higher effluent quality of MBR. Additionally, scenario SV1 has lower EP due to lower effluent volumes, leading to lower effluent loads if equal concentrations are assumed.

That issue is identified as a major drawback of this approach: with comparable effluent concentrations, a volume reduction in effluent automatically results in a lower EP, which is not supposed to be a realistic assumption. The treatment of concentrates should eventually lead to higher effluent concentrations compared to the treatment of diluted wastewater.

Overall, the adequate comparison of effluent loads or concentrations of the different water treatment processes can only be achieved if original effluent data from full-scale plants can be implemented. Both methodological approaches of this study to estimate effluent loads and resulting eutrophication have their drawbacks. However, the approach of estimating elimination via the ratio of influent to effluent loads (= variable concentrations) is considered a reasonable basis for a first assessment of EP in separation systems, if no suitable full-scale data is available.

5.3.8 NH_3 emissions during application of liquid fertilizer

The evaporation of nitrogen in the form of NH_3 during the application of liquid organic fertilizer is a major source for acidification in separation systems. However, the respective ratios of NH_3 emissions in relation to total applied nitrogen are estimated from pilot studies or data of comparable fertilizers. Emission factors for sewage sludge (8% of applied N), urine (10%) or digester sludge (stabilised: 6.3%, liquid: 22%) are estimated relatively high in comparison to mineral fertilizer (5%). If specific emission reduction techniques are applied (e.g. drag hoses, injection, instant ploughing after application), NH_3 emissions could be considerably reduced, resulting in lower acidification.

If NH_3 emissions during application of liquid organic fertilizer are reduced by 50% for each fertilizer, drawbacks in AP can be significantly reduced for separation scenarios (Figure 81). AP of urine separation scenarios SV and SC are decreased by 23%, and AP of vacuum scenarios without urine separation is even decreased by 92% in relation to the reference scenario. However, all separation scenarios are still significantly inferior in AP compared to scenario R.

Increased acidification in separation scenarios due to high NH_3 emissions during the application of liquid organic fertilizers has been identified as a major drawback for separation systems. Hence, emission reduction techniques should be implemented to minimize this drawback. Even with considerable efforts to minimize these emissions, separation systems most likely cause more emissions of NH_3 than mineral fertilizer application. The liquid form of organic fertilizers is seen as a major cause of this drawback, a fact that is inherent in the separation systems. The production of a solid fertilizer product would be an alternative.

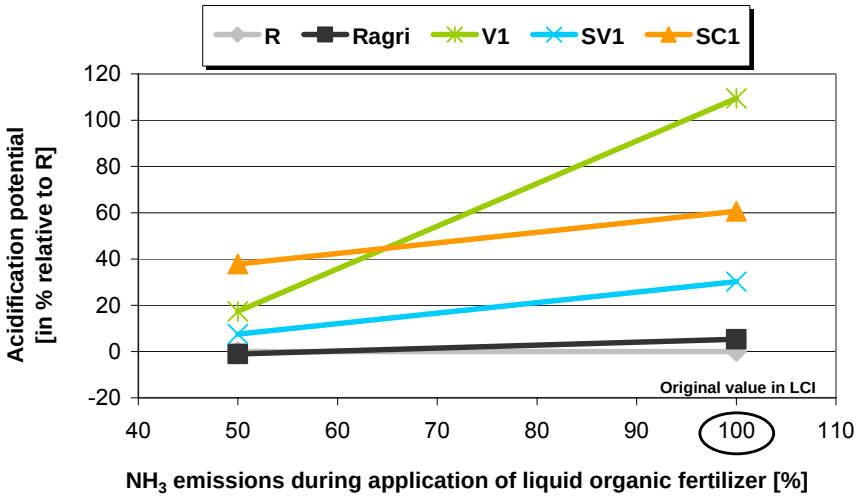

Figure 81: Variation of AP with NH_3 emissions during application of liquid organic fertilizer

5.3.9 Concentrations of Cu and Zn in drinking water

The contribution of drinking water to heavy metal loads in wastewater-derived fertilizers plays an important role for the two elements of Cu and Zn. Increased concentrations of these metals due to corrosion of pipe materials can lead to elevated concentrations in drinking water and organic fertilizers. In this study, concentrations of 0.16 mg/L Cu and 0.37 mg/L Zn are assumed, representing relatively high concentrations after contact with pipe materials.

If assumed concentrations of Cu and Zn are reduced by 50%, changes in freshwater aquatic ecotoxicity can be detected (Figure 82). While FAETP of separation scenarios with SBR or soil filter are decreasing by 1-3% in relation to the reference scenario, FAETP of reuse scenarios is slightly increasing (+1-3%). In this study, calculated concentrations of Cu and Zn in reused greywater are smaller than in drinking water, so that this advantage is partially cancelled for low concentrations of Cu and Zn in drinking water. The highest effect can be detected for scenario R_{agri}: the application of sewage sludge in agriculture transfers a high amount of Cu and Zn from drinking water to agricultural soil. Consequently, this drawback of sewage sludge application decreases with lower Cu and Zn content (-22% in FAETP for 50% lower Cu and Zn) and increases with higher Cu and Zn content (+30% in FAETP for doubling of Cu and Zn).

The transfer of Cu and Zn from drinking water to wastewater-derived fertilizers can contribute significantly to freshwater aquatic ecotoxicity, especially if sewage sludge is applied in agriculture. However, the qualitative comparison of separation scenarios with the reference system is not affected by assuming lower or higher concentrations of Cu and Zn in drinking water.

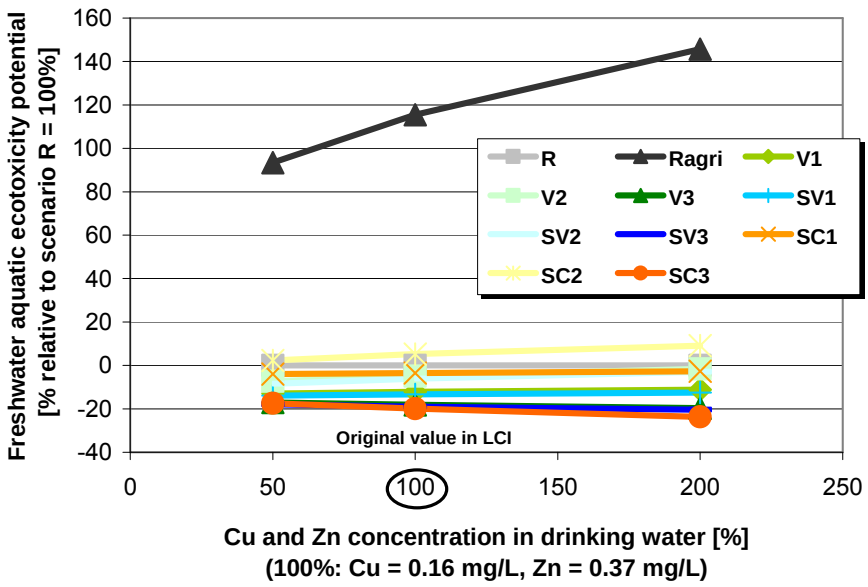

Figure 82: Variation of FAETP with concentrations of Cu and Zn in drinking water

5.3.10 Heavy metal data for mineral fertilizer

The content of heavy metals in mineral fertilizers (especially P fertilizer) is responsible for the high input of heavy metals in agricultural soil in the reference scenarios. Secondary fertilizers from urine and faeces have a lower heavy metal content, so that the separation scenarios are superior in terrestrial ecotoxicity to the conventional system. However, heavy metal data for mineral fertilizers used in the inventory relates to the year 1992. More recent data published in 2007 suggests a decreasing heavy metal content in mineral fertilizers, but this dataset is not representative due to a low number of samples (cf. annex 12.8.1).

If the 2007 dataset is used in the inventory, the benefits in TETP are slightly decreasing, but still stable for all separation scenarios (Figure 83). The relative decrease of the benefits of separation scenarios is due to a decreasing TETP of the reference scenario (-34%). A reduced content of toxic metals Cd, Cr, Ni, and Pb in mineral P fertilizer decreases the ecotoxicity potential caused by mineral fertilizer application. However, secondary fertilizers still contain less of these heavy metals.

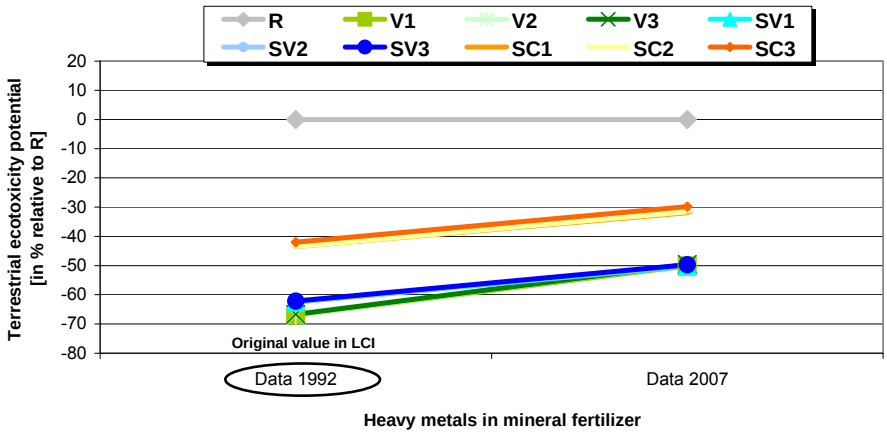

Figure 83: Variation in TETP with updated heavy metal data for mineral fertilizers

In general, the quality of mineral fertilizers seems to improve gradually in terms of heavy metal content, so that this specific benefits of secondary fertilizers (= lower heavy metal content) may diminish in the future. However, available resources of raw phosphate with low heavy metal content are limited (USGS, 2008). Hence, a sophisticated separation process is required to produce mineral P fertilizer with low heavy metal content, probably increasing resource demand and related emissions in the production process. If available, new and representative datasets for both the production of mineral fertilizers and their heavy metal content should be used to clarify this issue.

5.3.11 Plant availability of phosphorus in sewage sludge

Sewage sludge contains a major part of the wastewater-derived phosphorus. However, its plant availability is assumed to be limited (70%), basically due to the strong binding to ferric precipitates. Thus, the agricultural application of sewage sludge leads to a limited recycling of wastewater P to agriculture, so that mineral P fertilizer has to be amended to supply comparable P loads to agriculture. In consequence, scenario R_{agri} shows the highest human toxicity and terrestrial ecotoxicity of all scenarios.

If phosphorus in sewage sludge is estimated to be 100% plant available, TETP of scenario R_{agri} decreases significantly (-20%) in comparison to the reference system R (mineral fertilizer) (Figure 84). For human toxicity, the decrease is smaller (-15%), but it leads to a lower total HTP in relation to scenario R (Figure 85). This underlines the importance of estimating the correct plant availability of phosphorus in sewage sludge while assessing the human and ecotoxicological impact of sewage sludge application.

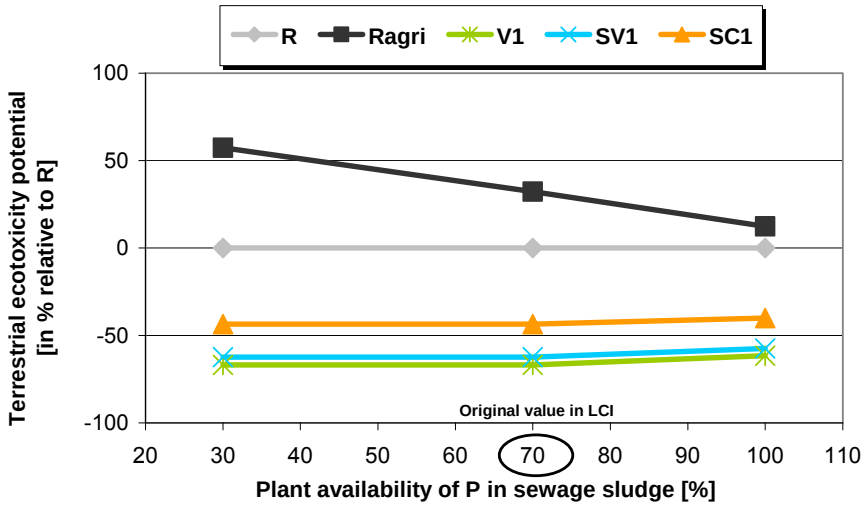

Figure 84: Variation of TETP with plant availability of phosphorus in sewage sludge

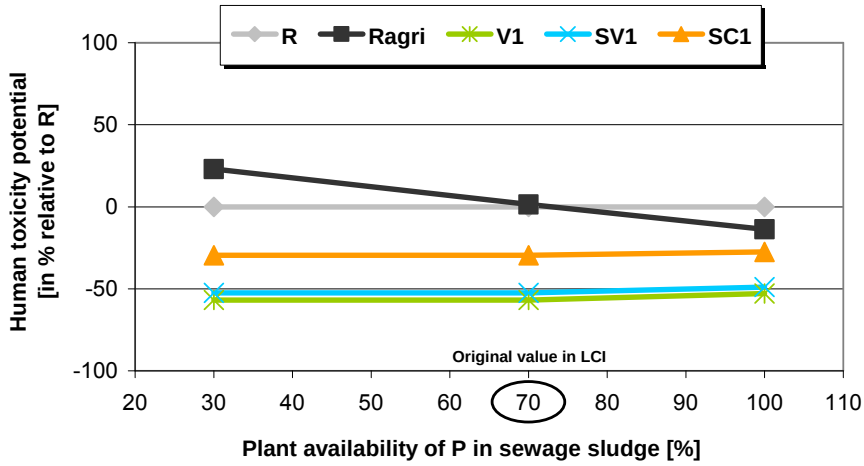

Figure 85: Variation of HTP with plant availability of phosphorus in sewage sludge

5.3.12 Alternative indicators for eutrophication

Eutrophication is an important impact category while assessing the environmental impacts of wastewater systems. However, calculation of the respective indicator can be performed following different methodologies, basically characterized by different impact factors for COD, nitrogen and phosphorus. To reveal possible influences of the indicator choice on the evaluation of this impact category, site-dependent characterization factors of EDIP 2003 (Hauschild and Potting, 2003) for Germany are used. Additionally, two different indicators are calculated: aquatic eutrophication (AEU) considering N and P emissions to surface waters (COD or TOC and N emissions to air are not included), and terrestrial eutrophication (TEU) considering only nitrogen emissions to air (NH_3, NO_x).

For aquatic eutrophication, the comparison between reference scenario and separation systems results in a similar trend than for the original indicator EP (cf. chapter 5.2.5). Soil filter scenarios are inferior in AEU (+ 47-205%), whereas vacuum scenarios with SBR or MBR are superior by 40-68% (Figure 86). Composting scenarios with SBR or MBR are comparable to the reference system. Thus, results of the new indicator AEU come to the same conclusion than EP, but the relative benefit or drawback of the respective separation scenario is more distinct. In other words, the relative benefits or drawbacks of separation scenarios are even more pronounced with this indicator. Hence, the indicator choice has no influence on the general findings in this impact category, but on their intensity.

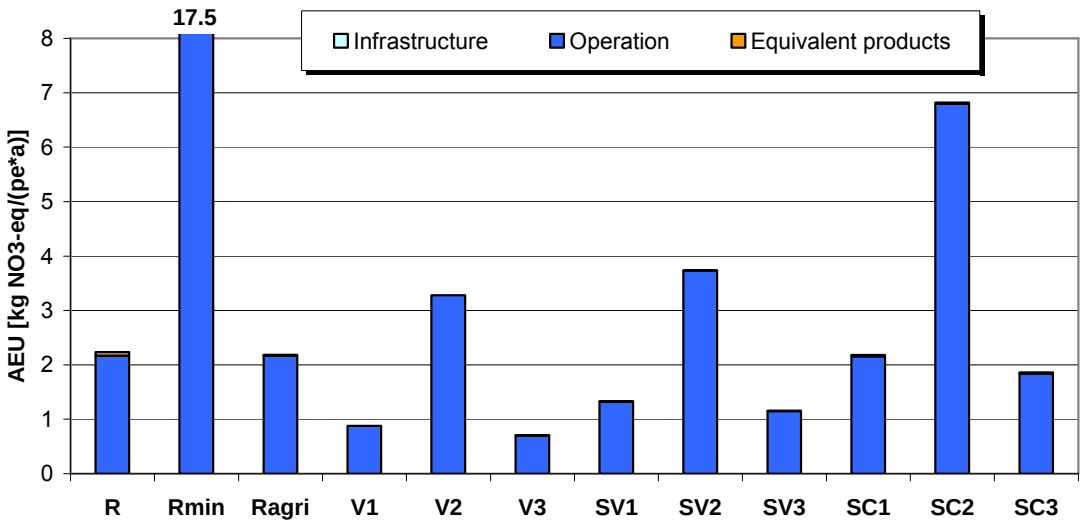

Figure 86: Aquatic eutrophication calculated with EDIP2003

For terrestrial eutrophication, the comparison tends to comparable results than the acidification indicator AP: all separation scenarios are inferior to the reference system, mostly due to high NH_3 emissions from application of liquid organic fertilizer (Figure

87). This fact of enhanced eutrophication of terrestrial ecosystems due to an increase in nitrogen emissions to air is not properly reflected in the original indicator of EP. Thus, the new indicator TEU has revealed another drawback of separation systems, i.e. the risk of enhanced eutrophication of terrestrial ecosystems. However, emission reduction techniques should be able to reduce this drawback of separation systems considerably by decreasing NH3 emissions in fertilizer application (cf. chapter 0).

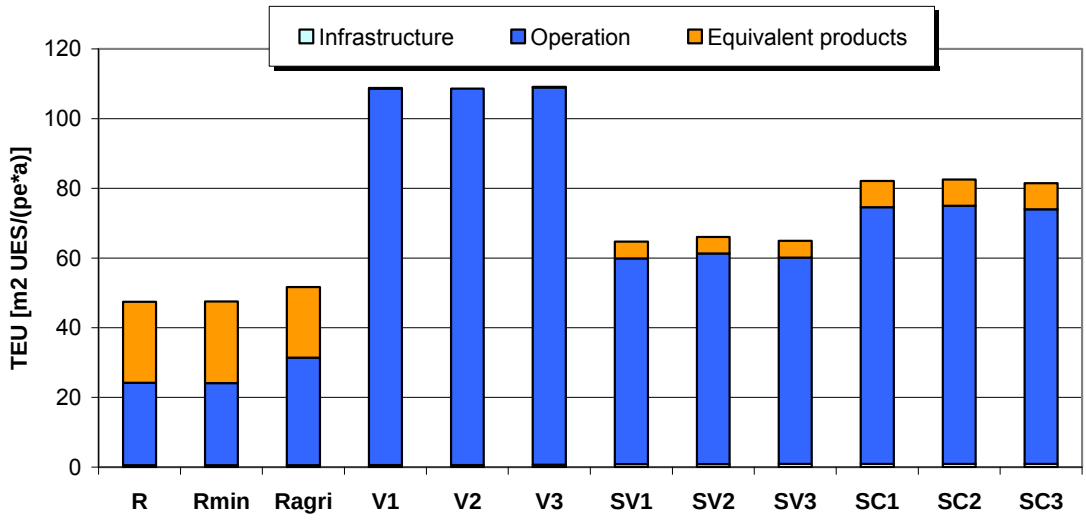

Figure 87: Terrestrial eutrophication calculated with EDIP 2003

5.3.13 Alternative indicators for ecotoxicity

The high uncertainty in calculating indicators for ecotoxicity has been addressed above (cf chapter 3.8.3). Especially the role of heavy metals in this assessment has to be evaluated with care. Two alternative indicators for ecotoxicity are calculated here to reveal possible influences of different approaches of toxicity assessment on the outcomes of this study, including aquatic ecotoxicity (AET) and terrestrial ecotoxicity (TET) following the methodology of IMPACT 2002+ (Jolliet et al., 2003a).

For aquatic ecotoxicity, the results are not comparable to the original indicator of FAETP: reuse scenarios are slightly superior to the reference system due to substitution of drinking water (decreasing Cu and Zn loads), while all other separation scenarios are inferior to scenario R with an increase in AET of 2-25% (Figure 88). This is particularly due to increased loads of Cu and Zn to agricultural soil (cf. chapter 5.1.5), because these emissions are characterized by high impact factors in the calculation of AET. The different characterization of aquatic ecotoxicity of Cu and Zn transferred to agricultural soil leads to a distinctively other result of the comparison in aquatic ecotoxicity depending on indicator choice.

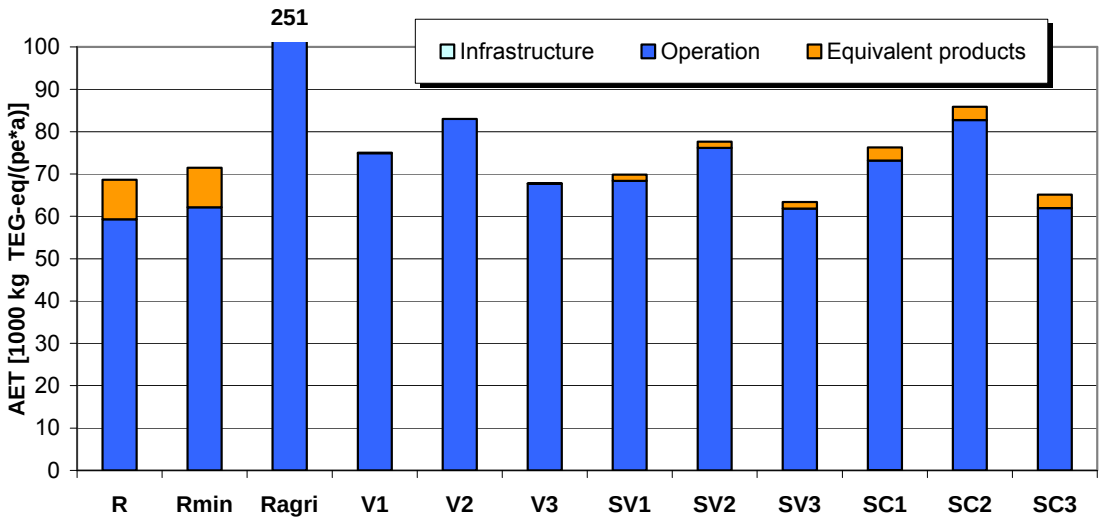

Figure 88: Aquatic ecotoxicity calculated with IMPACT 2002+

For terrestrial ecotoxicity, the differences between original and alternative indicator are even higher. While the original indicator TETP revealed significant benefits for all separation systems, the alternative indicator of TET calculates comparable or higher impacts in this category for the separation systems (Figure 89). Again, the role of Cu and Zn is crucial: high loads of these metals to agricultural soil are responsible for the drawbacks of separation systems, even though all other heavy metal loads (Cr, Cd, Hg, Ni, Pb) are decreased significantly (cf. chapter 5.1.5).

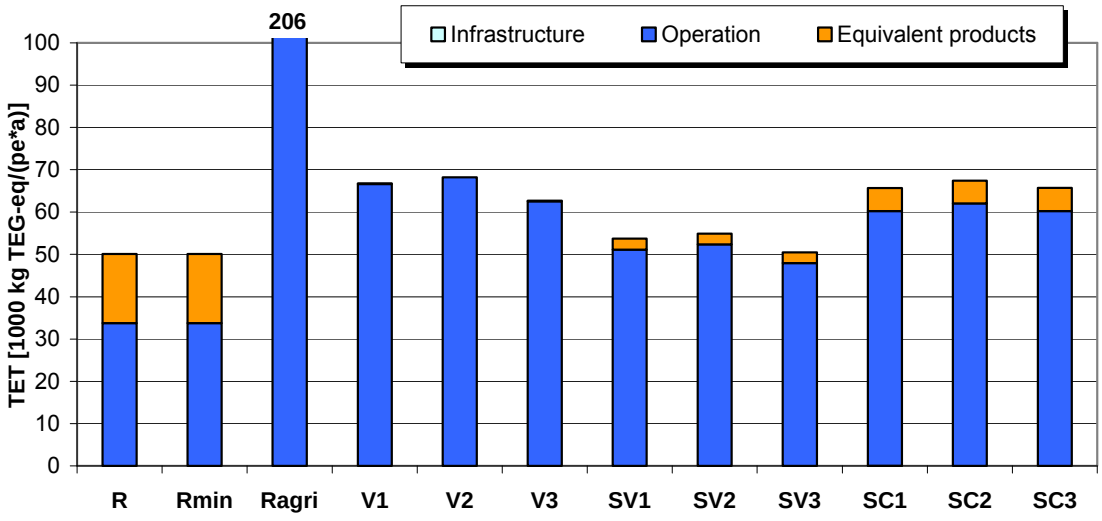

Figure 89: Terrestrial ecotoxicity calculated with IMPACT 2002+

Overall, it can be concluded that the assessment of ecotoxicity delivers different results in the comparison of reference and separation systems, depending on the respective indicator choice. Particularly the evaluation of Cu and Zn loads to agricultural soil is a crucial point here, because these heavy metals are contained in organic fertilizers from wastewater in high amounts. However, both metals belong to the group of essential trace metals, and their ecotoxicity is probably depending heavily on the specific conditions in the environment. Thus, the results of the ecotoxicity assessment from the original indicators of FAETP and TETP is taken as a basis for the comparison, even though these results should be seen with care before further knowledge in ecotoxicity assessment of metals is available.

5.3.14 Valuation with original UBA method

Valuation in this LCA study is based on an approach developed by UBA (Schmitz and Paulini, 1999) which is explicitly modified to include the indicators for toxicity. Thus, benefits of separation scenarios in toxicity indicators have a strong influence in valuation. However, the original UBA method has excluded these indicator categories on purpose due to methodical uncertainties in the calculation of the respective indicators.

If the original UBA method without toxicitiy indicators is used for valuation, the overall results of comparing each separation scenario with the reference system R are less significant. For example, the comparison between scenarios V1 and R results in a non-significant conclusion (Figure 90). The high acidification potential in scenario R more than offsets benefits in GWP and EP, and benefits in CED and ADP have another priority than AP, leading to a non-significant overall result by definition.

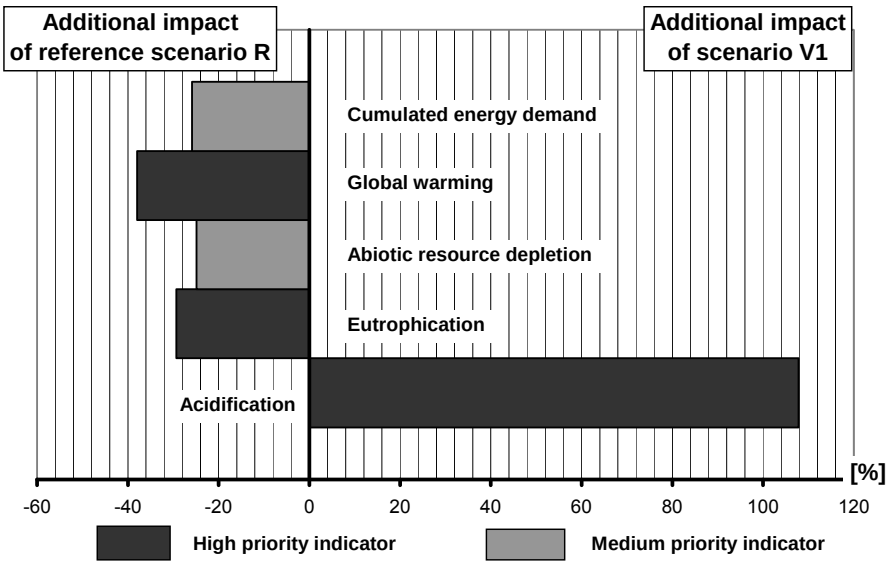

Figure 90: Comparison of scenario R and V1 with original UBA method in T diagram

In contrast, scenario SV3 remains significantly superior to the reference system applying the original UBA method (Figure 91). In total, only two separation scenarios (SV1 and SV3) remain significantly superior to the reference system using the original UBA method, whereas the results for all other separation scenarios are either non-significant or significantly worse (SC1 and SC3) than the reference system. The respective t-diagrams are shown in chapter 5.2.11 or in annex 12.10 (exclude toxicity indicators).

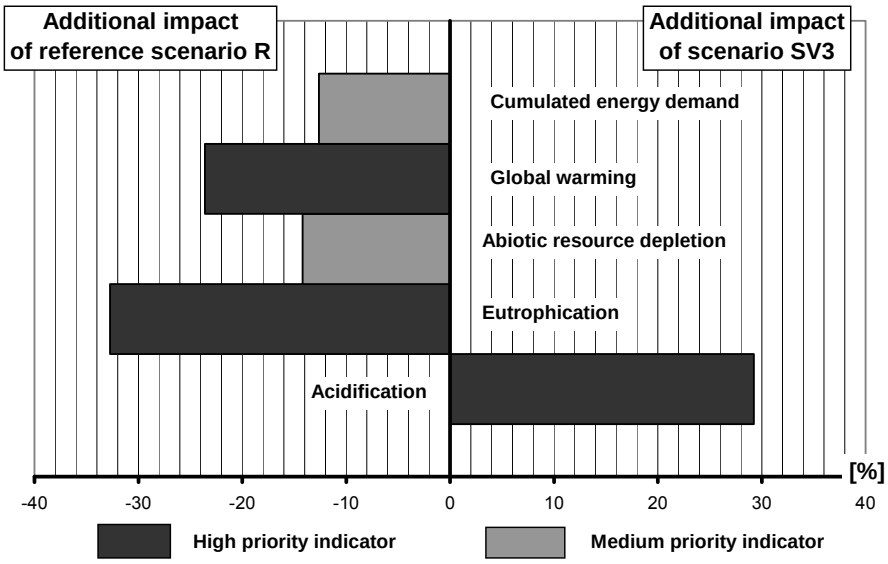

Figure 91: Comparison of scenario R and SV3 with original UBA method in T diagram

As a conclusion, it is a crucial decision for the overall result of this LCA whether to include or exclude the toxicity indicators in valuation. If toxicity indicators are included, more separation scenarios are rated better than the reference system. Due to the high uncertainties in the scientific basis for calculating toxicity indicators (cf. discussion in chapter 3.8.3), quantitative results of the toxicity assessment should be regarded with care. Nevertheless, stated benefits of separation systems in ecotoxicity are based on a lower input of toxic heavy metals into the environment. This benefit should in some way be included while comparing the scenarios. However, the quantitative uncertainty with toxicity indicators may lead to a biased valuation: a 50% drawback in acidification is then offset with a 50% benefit in TETP, although the calculation of the indicators is based on scientific models with different uncertainty. Thus, a valuation procedure including toxicity indicators is prone to a biased result until the impact models for toxicity are further improved.

6 Interpretation

The interpretation of the results of inventory analysis and impact assessment is the final step of an LCA. Relevant findings of LCI and LCIA are considered in relation to the specified goal of the study. Life cycle interpretation should reach conclusions, explain limitations and provide recommendations based on the results of the LCA (ISO 14040, 2006). The following chapter is divided into a discussion of relevant findings of this LCA, methodological issues and additional remarks.

6.1 Relevant findings of this LCA

For a comprehensive interpretation of the results of this LCA case study, findings of inventory analysis and impact assessment are summarized and discussed. As a starting point for the discussion, important results of the impact assessment are presented in two different schemes:

a) Comparison of separation systems with baseline scenario R

A matrix summarizes the benefits and drawbacks of separation systems for each indicator in relation to the baseline scenario R (Table 65). The simplified diagram accounts for the uncertainty of the LCIA results. Due to the large amount of inventory data, uncertainty is not quantified exactly in this study. However, the simplified presentation considers a defined uncertainty by postulating impact changes less than 10% as “not significant”, changes of 10-25% as “slight”, and changes greater than 25% as “significant”. Even though these numbers are set arbitrary, they are supposed to represent a rather conservative approach in accounting for uncertainty in this LCA. For toxicity indicators with high uncertainty in impact modelling, changes up to 50% are defined as “not significant”. The stability of the respective indicator results in sensitivity analysis is indicated by a background color (green = stable, yellow = partially unstable).

b) Synopsis of relevant findings in LCIA

For each impact category, a verbal summary lists the decisive processes, emissions and resources determining the indicator results (Table 66). Additionally, specific benefits and drawbacks of separation systems are mentioned in keywords. It has to be noted though that this verbal summary is by definition a synopsis of the results of this LCA. The reduction of relevant findings to certain keywords should not lead to a “shortcut” conclusion of this study.

Table 65: Matrix of indicator comparison between separation systems and reference scenario R

Scenario	V1	V2	V3	SV1	SV2	SV3	SC1	SC2	SC3
Faeces treatment	|——— Digestion ———|						|——— Composting ———|		
Urine treatment	|——— Digestion ———|			|——— Separation ———|					
Greywater treatment	SBR	Soil filter	MBR + reuse	SBR	Soil filter	MBR + reuse	SBR	Soil filter	MBR + reuse
Cumulative energy demand	++	++	+	+	+	+	0	0	-
Global warming	++	++	++	++	++	+	0	0	0
Abiotic resource depletion	++	++	+	+	++	+	0	0	-
Acidification	– –	– –	– –	– –	– –	– –	– –	– –	– –
Eutrophication	++	– –	++	++	– –	++	0	– –	0
Human toxicity*	+	+	+	+	+	+	0	0	0
Aquatic ecotoxicity*	0	0	0	0	0	0	0	0	0
Terrestrial ecotoxicity*	+	+	+	+	+	+	0	0	0

Note: indicator is ++ = better than 25% + = better than 10% 0 = comparable (± 10%) – = worse than 10% – – = worse than 25%

* for toxicity indicators: + = better than 50% 0 = comparable (± 50%) – = worse than 50%

Stability of results in sensitivity analysis: **GREEN = STABLE** **YELLOW = PARTIALLY UNSTABLE**

Table 66: Synopsis of relevant findings in Life Cycle Impact Assessment

Indicator	Decisive processes	Important emissions or resources	Benefits of separation systems	Drawbacks of separation systems	Remarks
CED	Greywater treatment, Faeces treatment, Biowaste?	-	Energy recovery from organic matter	Higher operational energy demand	Urine separation/ N fertilizer substitution/ GW reuse not decisive
GWP	Energy supply, Fertilizer application, Denitrification	CO_2, N_2O	Less N_2O from N fertilizer production and denitrification	More CH_4 from composting	–
ADP	Energy supply	Fossil fuels	Energy recovery from organic matter	Higher operational energy demand	Substitution of raw P ores has negligible influence
AP	Fertilizer application	NH_3	–	NH_3 emissions from fertilizer application and composting	(Emission factors estimated)
EP	Effluent of WWTP	P, N	Reduction of nutrient loads to WWTP	Insufficient P retention in soil filters, treatment of concentrates?	Depends on functional definition
HTP	Production and application of MF / SS	Fluoride, heavy metals	Substitution of MF / SS	–	MF and SS have comparable HTP
FAETP	Effluent of WWTP, Fertilizer application	Heavy metals	Substitution of MF / SS contaminated with HM, (GW reuse)	–	SS application causes highest FAETP
TETP	Fertilizer application	Heavy metals	Substitution of MF / SS contaminated with HM	–	–

MF: mineral fertilizer, SS: sewage sludge, GW: greywater, WWTP: wastewater treatment plant, HM: heavy metals

The following discussion is structured by the questions which were raised in the introduction of this thesis. Additionally, results are evaluated in relation to the data quality and the findings of sensitivity analysis.

1. Are separation systems more sustainable than the conventional system?

The comparison of environmental impacts between separation systems and the conventional approach reveals potential benefits in most of the impact categories (Table 65). Some indicators show distinct benefits or comparable impacts for separation scenarios, whereas other indicators reveal benefits for some scenarios, and comparable or even worse impacts for others:

- Human toxicity, terrestrial and aquatic ecotoxicity and global warming are comparable or lower for all separation scenarios.
- Cumulative energy demand, abiotic resource depletion, and eutrophication are lower, comparable, or higher depending on the layout of the separation system.
- Acidification is higher in all separation scenarios.

Regarding the different scenario groups of separation systems, which represent different options for handling of separated wastewater flows, the following trends in comparison to the reference system can be identified:

- Vacuum scenarios without urine separation (2-flow systems) have the lowest environmental impact scores of the three scenario groups. They are significantly better than the conventional system in many impact categories. In total, V scenarios have less environmental impacts than the reference system after grouping and weighting of the indicators if toxicity indicators are included in valuation.
- Vacuum scenarios with urine separation (3-flow systems) are also significantly better than the conventional system in some impact categories. In total, these scenarios are also superior to the reference system after grouping and weighting. However, their benefits are not as large as for the V scenarios, especially in terms of energy demand and resource depletion.
- Composting scenarios with urine separation (3-flow systems) have the highest environmental impact of all investigated separation systems. All indicator scores are comparable or worse than for the reference system. If indicators are grouped and weighted, the valuation result for the composting systems is either insignificant or worse than for the reference system.

In summary, it can be concluded from the results of this LCA case study that separation systems obviously can offer significant potentials for an increase in sustainability of wastewater management systems if compared to a conventional system. However, it requires a careful examination of system configuration and design to end up with a new sanitation system with less environmental impacts, especially if the conventional system is optimized in terms of nutrient removal (denitrification and P elimination) and energy demand (sludge digestion). The comparison between a conventional system with minimum standards (scenario R_{min}) and various separation systems shows a clear benefit for the separation scenarios. In other words, replacing a non-optimized conventional system with separation systems will most likely lead to a more sustainable solution, whereas the replacement of an optimized conventional system with separation systems is more difficult to evaluate. Nevertheless, the appropriate choice of configuration for a separation system should also lead to a more sustainable wastewater management in the latter case.

2. What are the decisive benefits of separation systems?

The decisive benefits of separation systems are – as expected – their improved capability to recover valuable resources from wastewater. The substitution of grid energy and mineral fertilizer is a crucial feature for the increased sustainability of separation systems. Another benefit is the reduced effort for the treatment of the remaining wastewater (= greywater) and presumably the better quality of the WWTP effluent. Below, the decisive benefits are discussed in detail.

a) Resource recovery: energy and nutrients

Energy recovery

Regarding energy recovery from organic matter, the energy content of faecal matter and urine alone (18-23 kWh/(pe*a)) does not justify the increased energy demand for its drainage with a vacuum system (15 kWh/(pe*a)) and digestion (8-11 kWh/(pe*a)). Only the addition of organic-rich household biowaste (+17 kWh/(pe*a)) to the digestion process results in an overall energetic benefit of the energy recovery option. Therefore, energy recovery from wastewater should definitely include the processing of biowaste. Alternatively, the energy demand of the vacuum system should be minimized by improved pump efficiency and system operation. With optimum energy recovery from wastewater and biowaste, separation systems require less energy if all relevant processes are considered in the system analysis.

Nutrient recovery

The nutrients which can be recovered from wastewater include the plant nutrients N, P, and K. Effective recovery rates are between 45-63% for nitrogen, 56-80% for phosphorus, and 54-78% for potassium depending on the system configuration of separation scenarios. The substitution of mineral fertilizers results in a reduction of environmental impacts by avoiding resource consumption and emissions during fertilizer production and by delivering fertilizers with better quality regarding their heavy metal content.

The environmental impact of mineral N fertilizer production is mainly associated with its energy demand, but the amount of substituted N fertilizer and its energetic equivalent do not play a major role for the energetic comparison of the different scenarios. Additionally, N_2O emissions from the production of mineral N fertilizer can be avoided and lead to a reduction in global warming potential. Overall, the substitution of mineral N fertilizer is not decisive for the environmental benefits of separation systems.

The main environmental benefit of nutrient recovery is associated with the substitution of mineral P fertilizer. On the one hand, the depletion of the finite resource of raw phosphate rock is reduced, and toxic emissions of fluoride and heavy metals during its exploitation can be avoided. On the other hand, mineral P fertilizer contains considerable amounts of heavy metals. Its substitution with organic fertilizer with low heavy metal content leads to a significant reduction of heavy metal inputs to agricultural soil. This benefit is also evident in comparison to the conventional method of phosphorus recovery via sewage sludge: phosphate fertilizers from separation systems are less contaminated with heavy metals than sewage sludge.

The substitution of mineral K fertilizer only results in marginal environmental benefits, because its production is not associated with relevant energy demand or emissions in this LCA.

b) Simplified treatment of remaining wastewater

Reducing energy demand for WWTP

Due to the low content of organic matter and nutrients, the remaining wastewater (= greywater) can be sufficiently treated with less energy demand than the combined wastewater. In case of an activated sludge plant (SBR), energy savings caused by a reduced aeration demand for carbon elimination and nitrification are relatively small (5-9 kWh/(pe*a)). Another suitable option is the low-energy treatment in a soil filter, which results in significant energy savings compared to the conventional system (22 kWh/(pe*a)). If faeces are composted, the co-treatment of nutrient-rich flushwater of gravity toilets offsets the benefits of simplified wastewater treatment considerably.

Improving WWTP effluent quality
Influent loads of nutrients and organic matter are considerably reduced for the wastewater treatment process in separation systems. Thus, effluent loads of N, P, and COD are expected to be lower in separation systems if the same process technology (activated sludge plant) is used than in the conventional system. However, the projected benefit of improved effluent quality has to be confirmed by more data from full-scale greywater treatment plants. The co-treatment of high strength flows with greywater (e.g. faeces filtrate) partially offsets this benefit. Greywater treatment in soil filters does not lead to a better effluent quality due to their limited potential for nutrient removal.

Overall, decisive benefits arise from energy recovery from wastewater and biowaste, substitution of mineral P fertilizer, and reduced energy demand for treatment of the remaining greywater. Minor benefits can be attributed to the substitution of mineral N and K fertilizer and the improved effluent quality of the wastewater treatment process.

Contrary to expectations prior to this LCA, the following issues have been identified as being not decisive for the environmental comparison of conventional and separation systems:

- The increased expenditures for the infrastructure of separation systems (especially for multiple drainage pipes) lead to a higher cumulative energy demand for their construction. However, this additional burden is clearly offset by energetic benefits from system operation. Thus, it is recommended to include the infrastructure in a simplified LCA, but with a simplified inventory which is limited to the sewer system. The contribution of infrastructure to total CED is less than 10%.
- The reduced water consumption with vacuum toilets does not lead to a substantial decrease in energy demand for separation systems. Likewise, the reuse of purified greywater for toilet flushing does not result in energetic benefits. An increased energy demand for treatment (MBR) and pumping energy for delivering greywater back to the households offset the energy savings in drinking water production. Substantial energetic benefits can presumably be realized if the volume of substituted drinking water is high (> 25 L/(pe*d)) and drinking water treatment is complex (~ 1 kWh/m³). It has to be noted here that a reduced demand of limited freshwater resources is not evaluated in the framework of this LCA (cf. chapter 6.3.1).
- The separation of urine does not lead to additional environmental benefits in this study if compared to vacuum scenarios without urine separation. Misdirected urine which is not properly separated in the toilet (30% of total

urine) leads to higher nutrient losses during treatment, so that the amount of substituted mineral fertilizer is smaller with urine separation. In terms of energy demand, low volume of nutrient-rich urine does result in lower fuel demand for transport, but these benefits are offset by lower biogas yield in digestion (3.5 kWh/(pe*a) due to loss of organic matter of separated urine for energy recovery) and additional energy demand for urine treatment (6 kWh/(pe*a) for abatement of organic micropollutants). However, it has to be noted here that the latter function is not included in vacuum scenarios without urine separation: the mixture of digested urine and faeces is not treated to remove micropollutants, mainly because an adequate process has not yet been successfully tested. Thus, the issue of micropollutant removal has been neglected in V scenarios.

3. Which environmental drawbacks of separation systems can be identified, and how could they be minimized?

In the course of this LCA, the following major environmental drawbacks of separation systems and possible options for their minimization could be identified:

NH_3 emissions during application of liquid organic fertilizer
The application of organic fertilizers will most likely lead to increased atmospheric emissions of NH_3. The liquid nature of the fertilizers (urine, digester residual) results in a high volatilization rate of nitrogen, decreasing the amount of available nitrogen and increasing the potential acidification of soils and surface waters via wet deposition of NH_3. Additionally, the input of excessive nitrogen can lead to eutrophication phenomena in terrestrial or marine ecosystems. Therefore, specific regulations should be established for the application of liquid organic fertilizers to minimize the losses of nitrogen, including mandatory low-emission application techniques (trail hoses, injection, or incorporation by ploughing) and rules for application (appropriate weather and soil conditions etc).

Phosphorus retention in soil filters
The long-term retention of phosphorus in soil filters is not sufficient to reach a reasonably low effluent concentration in greywater treatment. Phosphorus loads in greywater are substantial, so that total P effluent loads are likely to be increased if greywater treatment in soil filters is compared to conventional wastewater treatment. This drawback is especially important if nutrient-rich faces filtrate or sludge liquor from digestion is treated together with greywater. The implementation of an additional post-treatment stage for phosphorus removal (e.g. precipitation or adsorption on suitable filter media) should be seriously considered to prevent the increased eutrophication of surface waters.

Treatment of concentrates
The treatment of highly concentrated flows from processing of toilet wastewater (e.g. faeces filtrate in composting systems of sludge liquor from dewatering of digestor residual) should be separated from greywater treatment. The total amounts of nutrients and organic matter in concentrates can be considerable, deriving e.g. from urine which is not properly separated in the toilets. The mixing of these high-strength flows with greywater partially offsets the benefits of source separation. A separate treatment of concentrates with suitable technology (e.g. a high-load activated sludge process or membrane process) is strongly recommended, especially if soil filters are used for greywater treatment. Vacuum systems without urine separation do not produce concentrates, because the complete amount of toilet wastewater is applied in agriculture.

4. Which are the important key parameters for the impact assessment?

In the course of performing this LCA, key parameters for the impact assessment have been identified (Table 67). These parameters of the Life Cycle Inventory are found to be decisive for the results of this LCA and thus should be based on data with a sufficient quality. In particular, assumptions concerning these parameters should be justified carefully. In case of unavoidable uncertainty, sensitivity analysis should be performed to quantify the influence of variation of these parameters on the outcomes of the LCA. For a simplified LCA assessment of different separation scenarios, the listed parameters constitute the essential basis of a simplified inventory.

The listed key parameters of the inventory are not determined in a defined calculatory approach. A numerical analysis of all LCI parameters would be beyond the scope of this thesis. Identification of key parameters is rather based on experience from the setup of LCI, the impact assessment, and sensitivity analysis. Qualitative criteria for their selection are the significance or contribution of the related processes to certain impact categories and their relative importance for the conclusions of this LCA case study.

Table 67: Key parameters for LCI of separation systems

Process	Key parameters	Remarks
Wastewater flows	- Distribution of COD, N, P - Heavy metal content	Biowaste included?
Wastewater or greywater treatment	- Energy demand - Elimination of COD, N, P	Energy recovery from sludge digestion?
Vacuum system	- Energy demand	
Urine separation	- Separation efficiency - Energy demand of treatment	Micropollutant removal?
Digestion	- Energy demand - Quality of sludge liquor	Post-treatment of sludge?
Composting	- Quality of faeces filtrate	Solid-liquid separation?
Fertilizer application	- Plant availability of nutrients - NH_3 emissions	Nutrient equivalents of secondary fertilizers?
Mineral fertilizer	- Production data - Heavy metal content	

Note: The choice of key parameters is not based on a calculatory approach, but on experience gathered in inventory analysis, impact assessment and sensitivity analysis

Direct impacts versus indirect impacts

In general, direct emissions from core processes of wastewater treatment and fertilizer application are very important for the results of the impact assessment. Naturally, effluent emissions play a major role while evaluating the sustainability of wastewater treatment options. Additionally, fertilizer application has a significant environmental impact for separation systems due to the secondary functions of fertilizer supply. The impact categories of eutrophication, acidification, and ecotoxicity are determined by direct emissions to a large extent. The importance of direct emissions is confirmed by normalisation results where the respective indicators (EP, AP, TETP, FAETP) have a significant share to the total environmental burden in Germany (4-15%).

On the contrary, indirect environmental impacts from background processes (i.e. energy supply, transport, fertilizer production, supply of construction materials) are less significant for the comparison. Background processes mainly determine the energy demand (CED) and related emissions (GWP) or resource demand (ADP). Normalised results for these indicators show that the contribution of the investigated systems to total impacts in the respective impact categories is relatively small (< 3%). Nevertheless, the impact categories represent important environmental issues (i.e. the depletion of non-renewable energy resources and climate change) and should not be overlooked in the

comparison. Consequently, they are attributed a high relevance in grouping and weighting of indicators.

Data quality in relation to goal and scope

The evaluation of the quality of inventory data is an essential part of the interpretation of results in an LCA. The inventory data constitutes the basis of the LCA and consequently of its outcomes. Data quality should be evaluated in relation to the goal and scope of LCA, particularly concerning representativeness, consistency, completeness, and uncertainty (ISO 14044, 2006).

In this study, the quality of the LCI data has been discussed in detail in chapter 3.7.1. In summary, it has been found that the data quality of the inventory data used in this study can be described as sufficient in relation to the goal of this study. This thesis aims at a system analysis of prospective nature for the strategic planning of wastewater management options in the next decades. Thus, the mix of primary data from large-scale or pilot plants and secondary data from literature can be accepted for this case study, but has to be clearly communicated. For a case study targeting a decision support in a real situation, additional efforts should be invested to improve the data quality and thus make the LCA outcomes more defensible. Especially data from long-term operation of separation systems has to be generated to reach a sufficient level of accuracy for a decision support. Background data for materials, energy mix, transport and fertilizer production should be updated with more recent LCI datasets.

Sensitivity of results

The sensitivity of indicator results has been checked concerning the variation of important parameters of the Life Cycle Inventory and concerning the choice of indicators and valuation method. For the LCI parameters, the following conclusions can be drawn:

- The addition of biowaste into the digestion process is crucial for a net energetic benefit in the energy recovery scenarios.
- Transport distance of organic fertilizers is not decisive for CED and GWP of separation systems. Only if the complete toilet wastewater (= urine, faeces and flushwater) is transported over long distances (100 km in V scenarios), a significant influence can be detected (+13% in CED and +22% in GWP).
- Doubling the energy demand for urine treatment to 30 kWh/m³ is not decisive for the energetic comparison (+6% in CED).
- The energy demand of the vacuum system is a decisive parameter in the energy recovery scenarios. It should be minimized to yield a maximum energetic

benefit from energy recovery. An energy demand of more than 18 kWh/(pe*a) may offset benefits from biogas production.

- The separation efficiency of urine separation toilets is an influential parameter. Increasing the separation efficiency from 70 to 90%, several indicators are significantly influenced. Whereas GWP decreases by 4-5%, AP is either decreased by 11% (SV scenarios) or slightly increased by 3% (SC scenarios). EP decreases with increasing separation efficiency significantly for composting scenarios due to the reduction in nutrient loads to greywater treatment (co-treatment of faeces filtrate containing misdirected urine). EP reduction amounts to 29% for greywater treatment in SBR and 74% in soil filter.
- Increasing the volume of reused greywater to 50 L/(pe*d) results in marginal energetic benefits for the reuse scenarios (CED – 2-4%). Only if a complex drinking water treatment is assumed (1 kWh/m³), the energetic benefit can be substantial (CED – 5-9%).
- Assuming equal effluent concentrations for all activated sludge processes, the calculated EP for separation systems can increase considerably (+15-33%). The question of assuming variable or equal effluent concentrations for the different scenarios can be decisive for the comparison in EP.
- Minimizing NH_3 emissions during application of liquid organic fertilizers by 50% with adequate application techniques decreases the drawbacks in AP considerably.
- Increasing concentrations of Cu and Zn in drinking water have only marginal effects on the aquatic ecotoxicity of separation systems.
- An update of data for heavy metal content of mineral fertilizers still confirms benefits of secondary fertilizers in separation systems in terrestrial ecotoxicity.
- If P availability in sewage sludge is assumed to 100%, benefits of separation systems in HTP and TETP are still consistent if compared to scenario R_{agri}.

Overall, adopted LCI data evidently has an influence on specific indicators. However, just a few LCI parameters are found to be really influential on the results, whereas most of the other parameters do affect the results only marginal. Boundary conditions such as transport distance or heavy metal content of drinking water are less important for the comparison, whereas technical optimization of systems (e.g. separation toilets, vacuum system, or fertilizer application) should be seriously targeted to improve the sustainability of separation systems. Functional definitions can have a major influence on certain indicators (e.g. biowaste addition, effluent concentrations on EP) and should be carefully justified.

Concerning the sensitivity analysis for the indicator choice in LCIA, a twofold picture is evident. Some indicators are based on a widely accepted scientific basis, so that choosing a different LCIA model affects only the absolute scores, but not the qualitative results of the scenario comparison (e.g. aquatic eutrophication impact model). It has to

be noted that in case of aquatic eutrophication, the underlying model is fairly simple and may not be very precise in predicting actual impacts, but it is the only valid scientific model available. The splitting of eutrophication into separate indicators for aquatic and terrestrial effects leads to an additional indicator which reveals drawbacks for separation systems (increased terrestrial eutrophication). Here, the choice of an additional indicator can point out new hotspots and may influence the overall outcomes after grouping and weighting.

For other impact categories, various approaches of impact modelling are available which differ widely in the scientific basis for the calculation of indicator scores. The quantification of ecotoxicity can be performed with completely different approaches, and the indicator choice is highly relevant for the outcomes of the comparison. These limitations of the respective results (TETP, FAETP) should be clearly communicated here. Divergent results of other indicators can be additionally presented to underline the possibilities of a different outcome.

In total, the indicator choice has a strong influence on the outcomes in specific impact categories. Thus, it can seriously change the overall results of the comparison, especially after their evaluation via grouping and weighting. These issues underline the importance of further scientific improvement of impact assessment methods and probably the need for a generally accepted method for impact assessment. If the results of an LCA depend on the choice of LCIA method, the validity and relevance of its conclusions is obviously reduced.

Sensitivity analysis of the valuation method reveals a strong influence of the modification of the UBA method by including toxicity indicators. The high uncertainty of impact modelling demands a careful interpretation and thus may favour an exclusion of toxicity indicators from weighting. On the other hand, this would exclude a clear advantageous feature of separation systems from the overall evaluation, i.e. the superior quality of secondary fertilizers over their industrially produced equivalents. In the end, it is decided to include toxicity indicators in valuation with a distinct communication of the uncertainties involved.

6.2 Methodological issues

In the course of performing the present LCA case study, some methodological issues have been found to be important for further research in this area. In general, the structured approach of LCA is very helpful for a system analysis in wastewater management. Following the clearly defined framework of the ISO standard, a step-by-step procedure guides the researcher through all required parts of the assessment: functional definitions, inventory, impact assessment, and interpretation. At the same time, the ISO norm leaves enough room in its definitions to adapt the methodology closely to the specific needs and features of the respective case study.

However, the application of LCA in the field of strategic planning of wastewater management also implicates the acceptance of specific drawbacks which are connected to this methodology at present. Some of these issues can probably be overcome by a further progression in LCA development. In particular, the following methodological issues have been raised while applying LCA for wastewater management options, especially concerning the evaluation of new approaches such as source separation:

Goal and scope definition

- The definition of the primary system function should be as broad as possible to include all possible options for wastewater management.
- The careful consideration of all secondary functions of wastewater management is decisive for the comparison, preferably via system expansion.

Life Cycle Inventory

- The prospective nature of strategic studies naturally requires assumptions in the inventory data. Unavoidable high uncertainty in important parameters should be overcome by sensitivity analysis.
- Efforts in data collection should be focussed on improving data quality of key parameters, not on gathering detailed data for sub-processes of minor relevance. The quality of the results is presumably more influenced by the data quality of key parameters than by the completeness of the inventory. Key parameters of the LCI have been listed in chapter 6.1.
- The considerable effort for a detailed inventory of infrastructure in wastewater management is not justified by its relatively small impact on the results. A simplified inventory for infrastructure is recommended for strategic studies.

Life Cycle Impact Assessment

- The static nature of the substance flow model for the inventory and the scientific models for impact assessment prevents the evaluation of acute impacts of wastewater management. Effects of process dynamics (e.g. morning peak loads of nitrogen) which are important for WWTP layout, operation, and resulting effluent concentrations are not properly considered in LCA. Likewise, the effects of shock loads of emissions into aquatic ecosystems are not evaluated in LCIA.
- Specific effects of wastewater discharge on local ecosystems are not adequately evaluated in LCIA. Site-specific effect models are not available to predict the potential impacts of local emissions on a certain ecosystem (e.g. for eutrophication or ecotoxicity). Eventually, LCIA has to be connected to

available watershed models (e.g. MONERIS (Behrendt, 1999)) for a more precise prediction of impacts on local ecosystems.

- The assessment of human and ecotoxicity for important wastewater pollutants is connected to high uncertainty (e.g. heavy metals) or not available at present (e.g. organic micropollutants). Thus, the evaluation of these important environmental impacts of wastewater discharge is not based on a sufficient scientific basis to allow for a definitive statement.
- The evaluation of water use (i.e. water abstraction from a natural source) as an environmental impact is not yet included in LCIA. This impact category is essential for a comprehensive assessment of the ecological consequences of wastewater reuse. The implementation of water use into the LCIA framework is currently under development (see chapter 6.3.1 below for details).

Summarizing the issues above, it is obvious that Life Cycle Impact Assessment is still lacking a number of important features for a precise evaluation of the environmental impacts of wastewater management. However, the available LCIA methods are sufficient for a prospective system analysis in the author's opinion, targeting the comparison of different systems of wastewater management for strategic planning. If LCA is used as a decision support in a real case study, the impact assessment should be improved to include acute and local effects of wastewater discharge.

6.3 Additional remarks

6.3.1 Freshwater use

Freshwater is one of the most valuable resources, as it is an essential precondition for life and is not substitutable. For humans, it is indispensable as both drinking water and the basis for hygiene and food supply. On a global scale, many regions of the world suffer from limited or decreasing freshwater resource availability, with 900 million people around the globe lacking sufficient access to safe drinking water (WHO/UNICEF, 2008). Furthermore, the upcoming trends of urbanization, population growth and climate change are likely to intensify water stress on a regional scale. Therefore, a prospective sustainability analysis of water management systems should include an evaluation of the impact on freshwater resources.

However, traditional LCIA methods do not include an impact category for freshwater use. Only recently, the implementation of freshwater use at the level of LCI and LCIA is addressed in a joint effort (Koehler, 2008). Proposals of a methodology for inventory modelling, characterization to midpoint or even endpoint categories have been published (Frischknecht et al., 2008; Pfister et al., 2009), but are still being intensively

discussed in the scientific community. First attempts to implement the use of freshwater into the framework of LCA have addressed the following issues (Koehler, 2008):

- As water is an abiotic resource, three types of resources can be identified: deposits (non-regenerative within human lifetimes, i.e. fossil groundwater stocks), funds (regenerative stocks, i.e. aquifers and lakes), and flows (i.e. streams and rivers)
- Freshwater use can be divided into "utilization" (flows which are returned into the river basin where they have been abstracted) and "consumption" (ultimate withdrawal from a watershed, e.g. inter-basin transfer, evaporation, and incorporation into products).
- Impact assessment should evaluate both qualitative and quantitative issues. Water quantity should take into account the ratio between withdrawal and natural replenishment (water stress indicator). Chemical water quality impairments are already covered by LCIA (eutrophication, acidification, ecotoxicity), but other parameters (e.g. hygienic quality) are not yet characterized.

Hence, the use of freshwater resources cannot be evaluated within the LCA framework of this study. Nevertheless, separation scenarios offer the possibility of decreasing freshwater consumption in urban sanitation, either by decreasing toilet flushwater volume with vacuum toilets or by non-potable reuse of greywater. It is decided to quantify this potential for a decrease in freshwater consumption by showing the total daily water consumption per person for the different scenarios (Figure 92).

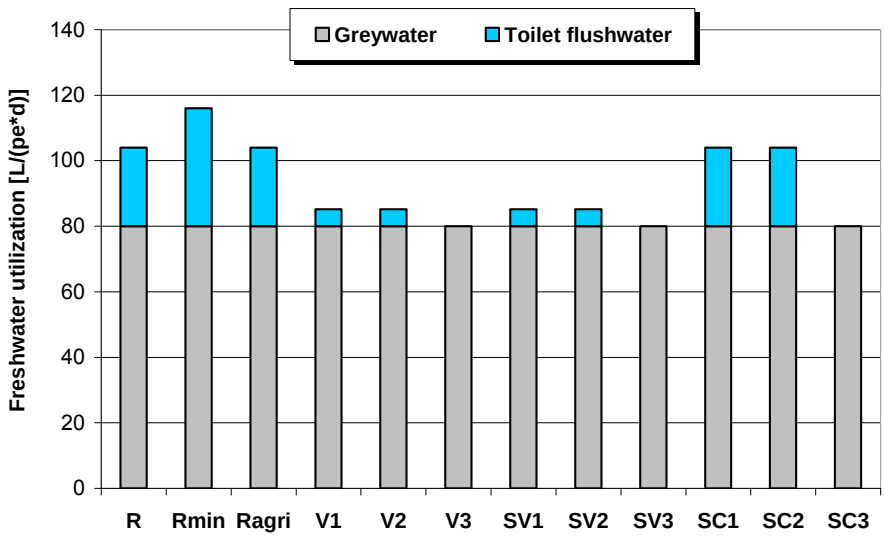

Figure 92: Freshwater utilization in each scenario

The potential for a reduction in freshwater utilization is limited with the separation systems investigated in this study: composting systems require the same amount of freshwater than the conventional system, whereas vacuum systems save around 19 L/(pe*d) of freshwater (-18%). Reuse scenarios substitute the complete toilet flushwater with purified greywater, so that freshwater demand is lowest in these systems. However, the largest part of the utilized freshwater is greywater (80 L/pe*d), and this fraction is not decreased by any scenario investigated in this study. The non-potable reuse of greywater for other purposes (e.g. garden irrigation, cleaning, washing etc) is an option to further reduce the freshwater demand substantially. Additionally, the amount of toilet flushwater used in conventional systems can be significantly higher than assumed in this study, depending on user behaviour and single flush volume. Thus, the reduction potential of vacuum toilets or greywater reuse for toilet flushing can be more important for the overall freshwater consumption, together with water saving appliances (washing machine, shower etc).

6.3.2 Economic sustainability of separation systems

Economic sustainability is one of the three integral parts of the concept of sustainability (WCED, 1987). The present work is limited to the comparison of the ecological impacts between the different sanitation scenarios with the methodology of Life Cycle Assessment. However, economic costs can be assessed with a similar concept which is called Life Cycle Costing (LCC). In analogy to LCA, LCC also takes into account all parts of the economic life cycle, i.e. construction, operation and maintenance, and (if applicable) disposal. Furthermore, the system boundaries can be broadened in LCC to include associated costs of other product systems.

Assessments of the economic sustainability of separation systems are difficult to find in the scientific literature. Reliable cost data of large-scale separation systems is not available, so that data from smaller pilot plants has to be used as a basis for cost projections. Lindert et al. estimated costs for a vacuum system for faeces drainage and found that the separation system can be economically beneficial if greywater can be treated in decentralized units and digested faeces can be disposed in agriculture (Lindert et al., 1997). Oldenburg and co-workers published cost calculations comparing different separation systems with a conventional system, stating that operational costs can be lower with separation systems, whereas total project costs are presumably comparable or higher (Oldenburg et al., 2007). Previous calculations of the same group resulted in economic benefits for separation systems over a 50 year period, but cost data was based on rough assumptions (Peter-Froehlich et al., 2004). Both publications stress the influence of local boundary conditions (e.g. population density, energy prices, water prices etc) on the results of the cost comparison. Dockhorn made a detailed cost analysis of a large urban sanitation system (350000 inhabitants) and identified economic benefits for separation systems (Dockhorn, 2007). These benefits derive from reduced costs for

the treatment of the remaining wastewater and saved costs for mineral fertilizer. Even for a gradual change of wastewater management systems towards source separation, economic incentives are existent which may stimulate further development and gradual implementation of separation systems.

Overall, the economic sustainability of separation systems has yet to be proven in a real full-scale case study, as reliable cost data for the new sanitation options is not available at the moment. However, projections have shown that separation systems have the potential to yield economic benefits by reducing costs for the wastewater treatment plant and by substituting mineral fertilizer. Rising prices for energy and fertilizer will certainly increase the economic benefits of resource recovery from wastewater in the future. At the moment, separation systems seem to be at least comparable to the conventional system regarding the total costs.

6.3.3 Social LCA

The third pillar of sustainability is related to the social impacts of a certain activity in society. However, consistent methodologies for societal Life Cycle Assessment are still under development (Jorgensen et al., 2008). Specific problems arise during the assessment of the social impacts of a certain product, e.g. how to establish a quantitative relation between indicators and functional unit, how to obtain regionalized inventory data, and how to quantify the impacts properly (Klopffer, 2008). Recently, the UNEP/SETAC initiative released a framework for social LCAs to establish mandatory guidelines (UNEP, 2009). Some studies have already tried to include the social aspects into their sustainability assessment of wastewater systems (Balkema, 2003). Suggested indicators in this study are cultural acceptance, level of required expertise, institutional requirements, and possibilities for participation. In all, more research is required to systematically assess the social impacts of different systems for wastewater management.

7 Conclusions

The target of increasing sustainability for all technical systems in society includes the field of urban wastewater management. The conventional system of combined wastewater drainage and treatment is characterized by a disposal-oriented approach, historically developed for the safe disposal of human excreta to prevent water-borne diseases. For the protection of receiving surface waters, combined wastewater is treated in activated sludge plants, thus consuming high amounts of energy and chemicals ("end-of-pipe" technology). Recently, new approaches for urban wastewater management have been described which aim at the recovery of valuable resources from wastewater. Low-volume, nutrient-rich flows of urine and faeces are separated at the source to facilitate the recovery of energy and nutrients, whereas the remaining greywater can be treated with less effort to reach sufficient effluent quality for discharge or even wastewater reuse. However, the environmental sustainability of these separation systems has to be systematically investigated to identify and quantify decisive benefits and drawbacks in comparison to the conventional approach. Therefore, the methodology of Life Cycle Assessment (ISO 14040/44) is applied in this thesis to compare the conventional and various separation systems for urban wastewater management in their environmental impacts.

Applying LCA methodology

The definition of system functions, boundaries and scenarios is based on previous research in this area and conforms to the ISO standards 14040/44. As a hypothetical case study, twelve different scenarios for the integrated management of urban wastewater and biowaste from 5000 inhabitants are investigated, including both infrastructure and operation of all relevant processes. Secondary functions of separation systems (supply of energy and nutrients) are considered by expanding the conventional system with the respective processes of energy and mineral fertilizer production. A detailed Life Cycle Inventory for all relevant processes is set up, describing the elemental flows through the system and related emissions and resource demand. Inventory data of core processes is based on data from pilot projects, literature and qualified assumptions, whereas background processes (energy supply, transport etc) are described with available process modules from databases. A sophisticated substance flow model is developed using the LCA software UMBERTO®. Finally, aggregated resource demand and emissions are evaluated in Life Cycle Impact Assessment. Eight indicators are calculated which describe the environmental impacts of fossil energy demand, global warming, abiotic resource depletion, eutrophication, acidification, and human and ecotoxicity.

Findings of the LCA case study

The results of the impact assessment show that separation systems offer potentials for a reduction in environmental impacts of urban wastewater management. The energy recovery from organic matter of faeces, urine, and household biowaste via digestion in a biogas plant can decrease the cumulative energy demand by up to 40% and related emissions of greenhouse gases by up to 46%. Energetic benefits of substituting mineral fertilizer with recovered nutrients are relatively low, but the quality of the organic fertilizer is significantly better: heavy metal content of mineral P fertilizer is relatively high, and its substitution with organic P fertilizer from urine and faeces leads to a reduction of heavy metal input on agricultural soils. This benefit is consistent if separation systems are compared to agricultural application of sewage sludge, the conventional way of nutrient recycling.

The reduction of energy demand for the WWTP is relatively small if comparable technology (activated sludge) is applied in greywater treatment. With natural treatment in soil filters, this energy demand can be considerably decreased. However, insufficient retention of phosphorus in soil filters can lead to an increased eutrophication potential (+ 40-140%) of receiving surface waters. With activated sludge technology, effluent quality of the WWTP is comparable or better than in a conventional system, resulting in comparable or less eutrophication potential and aquatic ecotoxicity. Using membrane bioreactors, greywater can be easily treated for non-potable reuse to reduce freshwater demand. Energetic benefits of wastewater reuse can only be detected if drinking water treatment is complex and reuse volume is high.

Serious drawbacks of separation systems are the increased emission of ammonia during the application of liquid organic fertilizers, leading to increased acidification (+ 30-110%) and eutrophication of terrestrial ecosystems. Concentrated flows such as faeces filtrate or sludge liquor should be treated separately and not together with greywater, as they contain considerable amounts of nutrients. Soil filters for greywater treatment should be equipped with an additional treatment stage for phosphorus removal. Each of these drawbacks is not system-inherent, but can be overcome by suitable improvements of technology.

If indicator results are summarized by grouping and weighting, separation systems can offer significant potentials for an overall increase in environmental sustainability of wastewater management. However, the choice of an appropriate combination of process technology is essential for the realization of these potentials, because the conventional system has already been optimized in terms of energy demand and effluent quality in the last decades. The higher demand for infrastructure (multiple pipe networks) in separation systems is not decisive for the environmental comparison.

Validity and robustness of results

Data quality of the present LCA can be described as sufficient for a prospective study of strategic planning and evaluation of wastewater management options. For a decision support in a real case study, data quality of key parameters should be improved to end up with more defensible results, and background data should be updated. Respective key parameters of the inventory have been identified in this thesis to facilitate the data acquisition in future LCA studies of wastewater management.

Sensitivity analysis revealed that only a few inventory parameters have a decisive influence on the results in this LCA. Whereas boundary conditions have only marginal effects on indicator results, specific process parameters (e.g. energy demand for vacuum plant, urine separation efficiency, techniques for application of organic fertilizer) may heavily influence the results and thus offer major potentials for system optimization. Functional definitions and the choice of scientific models for calculation of indicators have a distinct influence on the comparison and should therefore be carefully justified.

Methodological issues of LCA in wastewater management

In general, the structured approach of LCA is apparently valuable for a systematic assessment of the environmental impacts of wastewater management systems. Following the step-by-step procedure of the ISO standard and adapting it to the specific case study, a detailed system analysis is possible which draws the attention to the decisive issues of the comparison. Thus, the identification of important environmental benefits and drawbacks of the different scenarios as well as potentials for further process optimization is easily achievable. This holds especially true if scenarios with different underlying principles (combined vs. separation systems) are compared in their sustainability. Therefore, the life cycle approach is definitely recommended by the author for further studies in this area.

Life Cycle Assessment in its current state is suitable for the tasks described above. However, the available methods of impact assessment do not consider spatial or temporal dynamics of emissions, so that the actual impacts of effluent discharge in a local ecosystem may not be adequately characterized with the presented indicators. Additionally, an indicator for freshwater withdrawal from natural sources should be included in the impact assessment to account for a major environmental impact in water-scarce regions, especially concerning the evaluation of wastewater reuse scenarios. The further development of existing and new indicators for Life Cycle Impact Assessment is a major task for the LCA community.

Outlook on further research

Increasing sustainability of our technical systems in society is a major challenge of the near future. As shown in this thesis, new approaches in wastewater management can obviously contribute to this task by changing the functions of the existing systems from disposal and end-of-pipe technology to separation and recovery of resources. This requires a fundamental change of our perception of wastewater, away from a waste that has to be disposed to a resource that can be exploited. Comparable transitions to a new perception have been successfully implemented in solid waste management in the past. The technology for a change in wastewater management is available today and will be further optimized in the future. However, it is important to carefully evaluate the new approaches in terms of their sustainability, including environmental, economic and social impacts. Only if all three pillars of sustainability are adequately considered, the new approaches can prove that they are superior to the old concepts.

Therefore, more research is required in the field of sustainability assessment of wastewater systems. Methods for sustainability assessment have to be improved to deliver a more realistic picture of the true impacts of wastewater management on the whole water cycle, including the production of drinking water and other functions of natural water resources. Following the methodological approach of this thesis, more case studies should be carried out to evaluate the benefits of separation systems in specific situations, with local boundary conditions and specific technologies available in different countries. Only then, society will gradually change its picture of wastewater management from a 19th century concept of urban hygiene to a 21st century concept of resource recycling.

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9 List of Abbreviations

ADP	Abiotic Depletion Potential
AET	Aquatic Ecotoxicity
AEU	Aquatic Eutrophication
AP	Acidification Potential
CAS	Conventional activated sludge
CED	Cumulative Energy Demand
COD	Chemical Oxygen Demand
EP	Eutrophication Potential
FAETP	Freshwater Aquatic Ecotoxicity Potential
GWP	Global Warming Potential
HTP	Human Toxicity Potential
ISO	International Standardization Organisation
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MBR	Membrane Bioreactor
SBR	Sequencing Batch Reactor
TETP	Terrestrial Ecotoxicity Potential
TET	Terrestrial Ecotoxicity
TEU	Terrestrial Eutrophication
TOC	Total Organic Carbon
WWTP	Wastewater Treatment Plant

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Contents

12.1 Composition of partial flows of water, wastewater and waste

The following tables list the results of an extensive literature research concerning the average composition of human urine (Table 68) and faeces (Table 69), greywater from kitchen, bath and washing machine (Table 70), biowaste from households and loppings (Table 71).

Table 68: Volume flow and composition of human urine

Parameters	unit	1)	2)	3)	4)	5)	6)	7)	8)	9)	This study
Volume flow	L/(pe*d)	1.37	1.73		1.57				1.20		**1.5**
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	11.5	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**
K	g/(pe*d)	1.94	2.32	2.70	2.60			2.25	2.20		**2.6**
Na	g/(pe*d)		2.81	3.50			6.00				**3.5**
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				**4.8**
S total	g/(pe*d)			1.32				0.63			**0.8**
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**

1) Lange and Otterpohl, 2000: collection of literature values
2) Jönsson et al., 1997: average values after storage (14 d)
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 69: Volume flow and composition of human faeces

Parameters	unit	1)	2)	3)	4)	5)	6)	7)	This study
Volume flow (wet mass)	kg/(pe*d)	0.14	0.14	0.22	0.13		0.18		**0.14**
Dry matter	g/(pe*d)	35.11		50.96	21 – 34		44.70		**45**
Organic dry matter	g/(pe*d)	29.23			ca. 18		44.80		**42**
COD	g/(pe*d)	78.27		4.57		37.00	33.00		**35**
BOD	g/(pe*d)			3.35		19.00	11.10		**14**
TOC	g/(pe*d)		46.58				21.40		**21**
N total	g/(pe*d)			1.95	1.80	1.90	2.00		**1.5**
P total	g/(pe*d)	0.96	0.55	0.68	0.31 – 0.77		0.70		**0.5**
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**
Mg	g/(pe*d)				0.24				**0.2**
Cl	g/(pe*d)				0.06	0.10			**0.06**
S total	g/(pe*d)			0.21					**0.2**
Cu	mg/(pe*d)			1.74	1.96			1.10	**1.5**
Zn	mg/(pe*d)			46.41	5.1 – 10.3			10.80	**10**
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	**0.2**
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**

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)
4) Ciba-Geigy, 1977
*5) Koppe and Stozek, 1999: calculation assuming 150 g faeces/(pe*d)*
6) Oldenburg, 2002: average values, including toilet paper
7) Vinneras, 2001

Table 70: Volume flow and composition of greywater from households

Parameters	unit	1)	2)	3)	4)*	5)	6)	7)	8)	9)	This study	This study*
Volume flow	L/(pe*d)	110.00				82.19	70.79	69.00			**80.00**	**80.00**
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**

**) without loads from tap water*
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 (recalculated)
7) Bahlo, 1999
8) Butler et al., 1995: average values from the U.S. 1974-1986
9) Raach et al., 1999 (calculated)

Table 71: Mass flow and composition of kitchen and garden biowaste

Parameters for kitchen biowaste	unit	1)	2)	3)	4)	5)	6)	This study
Mass flow	kg/(pe*d)	0.16	0.18			0.18	0.20	**0.20**
Dry matter	g/(pe*d)	50.37					32.00	**50.00**
Organic dry matter	g/(pe*d)	35.81		49.01	30.80	46.09	27.20	**36.00**
TOC	g/(pe*d)	13.16			10.90		13.60	**13.00**
N total	g/(pe*d)	0.82	0.89		1.00		0.43	**0.90**
P total	g/(pe*d)	0.20	0.16	0.17	0.13		0.18	**0.20**
K	g/(pe*d)	0.60	0.50				0.28	**0.60**
Na	g/(pe*d)					6.729		**1.20**
Ca	g/(pe*d)	0.98					0.35	**1.00**
Mg	g/(pe*d)	0.22					0.03	**0.22**
Cl	g/(pe*d)					10.37		**3.00**
S total	g/(pe*d)		0.09					**0.10**
Cu	mg/(pe*d)	1.06	0.44				0.34	**1.00**
Zn	mg/(pe*d)	7.28	1.85				0.98	**7.30**
Cd	mg/(pe*d)	0.01	0.007				0.00	**0.01**
Ni	mg/(pe*d)		0.24				0.30	**0.20**
Hg	mg/(pe*d)	0.01	0.01				0.00	**0.01**
Pb	mg/(pe*d)	1.06	0.11				0.06	**0.60**
Cr	mg/(pe*d)		0.50				0.05	**0.50**

Parameters for garden biowaste	unit	1)	7)	8)	This study
Mass flow	kg/(pe*d)	0.33			**0.30**
Dry matter	g/(kg wet mass)	424.62	400.00		**410.00**
Organic dry matter	g/(kg d.m.)	72.29	70.00		**710.00**
TOC	g/(kg d.m.)	391.22	348.60		**370.00**
N total	g/(kg d.m.)	13.93	12.00	6.80	**11.00**
P total	g/(kg d.m.)	9.00	5.28		**5.00**
K	g/(kg d.m.)	12.24	14.94		**13.60**
Na	g/(kg d.m.)				**0.20**
Ca	g/(kg d.m.)	21.95	44.02		**33.00**
Mg	g/(kg d.m.)	3.87	4.80		**4.50**
Cl	g/(kg d.m.)			< 0.3	**0.30**
S total	g/(kg d.m.)			0.50	**0.50**
Cu	mg/(kg d.m.)	27.82	10.00	10.00	**19.00**
Zn	mg/(kg d.m.)	152.82	60.00	57.00	**110.00**
Cd	mg/(kg d.m.)	0.60	0.27	< 2.2	**0.40**
Ni	mg/(kg d.m.)		3.70		**3.70**
Hg	mg/(kg d.m.)	0.15	0.23		**0.20**
Pb	mg/(kg d.m.)	47.27	4.80	< 6.6	**4.80**
Cr	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

12.2 Characterization factors for Life Cycle Impact Assessment

The following tables list the characterization factors of Life Cycle Impact Assessment for all calculated indicators

Table 72: LCIA characterization factors for emissions to air: GWP, AP, EP, TEU, AEM

Name	**GWP**	**AP**	**EP**	**TEU**
Indicator	Global Warming Potential (100a)	Acidification Potential	Eutrophication Potential	Terrestrial eutrophication
Source	[1]	[2]	[3]	[4]
Unit	kg CO_2-eq/kg	kg SO_2-eq/kg	kg PO_4-eq/kg	m² UES/kg
CH_4	23			
CO_2 (fossil)	1			
HCl		0.88		
HF		1.6		
H_2S		1.88		
NH_3		0.77	0.11	80
N_2O	296			
NO_x		0.19	0.05	23.6
SO_2		0.6		

[1]: IPCC, 2001 (related to a time horizon of 100 years)
[2]: Guinée et al., 2002 (including site-dependent fate and effect factors for NO_x, NH_3, SO_2, scenario OA 1995 for Germany)
[3]: Huijbregts and Seppala, 2001 (including fate and effect factors for Europe)
[4]: Hauschild and Potting, 2003 (site-dependent factors for Germany)

Table 73: LCIA characterization factors for emissions to air: HTP, FAETP, TETP, AET, TET

Name	**HTP**	**FAETP**	**TETP**	**AET**	**TET**
Indicator	Human Toxicity Potential	Freshwater Aquatic Ecotoxicity Potential	Terrestrial Ecotoxicity Potential	Aquatic Ecotoxicity	Terrestrial Ecotoxicity
Source	[1]	[1]	[1]	[2]	[2]
Unit	kg 1,4-dichlorobenzene-eq/kg			kg of triethylene-glycol-eq/kg	
Particles	8.20E-01				
Dust (PM10)	8.20E-01				
Sb	6.71E+03	3.72E+00	6.11E-01	2.96E+05	2.01E+04
As	3.48E+05	4.95E+01	1.61E+03	5.48E+04	4.19E+05
Be	2.27E+05	1.71E+04	1.77E+03		
Pb	4.67E+02	2.40E+00	1.57E+01	4.01E+04	1.31E+05
Cd	1.45E+05	2.89E+02	8.12E+01	4.28E+05	9.12E+05
Cr^{III}	6.47E+02	1.92E+00	3.03E+03		
Cr^{VI}	3.43E+06	7.69E+00	3.03E+03		
Cr^{III+VI}	3.49E+04	1.98E+00	3.03E+03	6.70E+04	3.82E+05
Co	1.75E+04	6.39E+02	1.09E+02		
Cu	4.30E+03	2.22E+02	6.99E+00	2.94E+06	1.18E+06
Ni	3.50E+04	6.29E+02	1.16E+02	1.79E+05	5.63E+05
Hg	6.01E+03	3.17E+02	2.83E+04	7.86E+05	3.84E+06
Se	4.77E+04	5.46E+02	5.35E+01	4.78E+05	9.32E+03
Tl	4.32E+05	1.55E+03	3.40E+02		
V	6.24E+03	1.73E+03	6.65E+02	2.04E+05	
Zn	1.04E+02	1.78E+01	1.20E+01	2.04E+05	1.01E+06
Sn	1.73E+00	2.54E+00	1.44E+01		
Benzo(a)pyrene	5.72E+05*	8.78E+01	2.41E-01	2.50E+04	2.42E+01
Benzene	1.90E+03	8.37E-05	1.56E-05	2.44E-02	4.72E-03
Chlorobenzene	9.23E+00	4.68E-04	7.29E-04	5.07E+00	3.36E-01
Formaldehyde	8.31E-01	8.26E+00	9.40E-01	1.67E+00	2.54E+00
PAH w/o B(a)P	5.72E+05	1.72E+02	1.02E+00		
PAH, unspecified	5.72E+05	1.72E+02	1.02E+00		
PCDD, PCDF	1.93E+09	2.13E+06	1.20E+04	3.94E+05	2.59E+02
NH_3	1.00E-01			3.90E+00	9.80E+00
HCl	5.00E-01				
HF	2.85E+03	4.64E+00	2.95E-03	7.84E-01**	1.79E+02**
NO_x	1.20E+00				
SO_2	9.60E-02				
H_2S	2.20E-01				

** factor for PAH w/o benzo(a)pyrene is adopted*
*** calculated from fluorene*
[1]: Huijbregts et al., 2000
[2]: Jolliet et al., 2003b

Table 74: LCIA characterization factors for emissions to soil: HTP, FAETP, TETP, AET, TET

Name	**HTP**	**FAETP**	**TETP**	**TET**	**AET**
Indicator	Human Toxicity Potential	Freshwater Aquatic Ecotoxicity Potential	Terrestrial Ecotoxicity Potential	Terrestrial Ecotoxicity	Aquatic Ecotoxicity
Source	[1]	[1]	[1]	[2]	[2]
Unit	kg 1,4-dichlorobenzene-eq/kg			kg of triethylene-glycol-eq/kg	
As	3.18E+04	1.34E+02	3.34E+03	2.43E+06	3.87E+05
Pb	3.28E+03	6.53E+00	2.34E+01	7.54E+05	2.64E+05
Cd	1.96E+04	7.76E+02	1.67E+02	5.28E+06	2.91E+06
Cr^{III}	5.13E+03	5.25E+00	6.30E+03		
Cr^{VI}	8.55E+03	2.10E+01	6.30E+03		
Cr^{III+VI}	5.16E+03	5.41E+00	6.30E+03	2.25E+06	4.49E+05
Co	2.39E+03	1.71E+03	2.23E+02		
Cu	9.39E+01	5.95E+02	1.44E+01	6.92E+06	2.04E+07
Ni	2.68E+03	1.69E+03	2.39E+02	3.30E+06	1.26E+06
Hg	5.92E+03	8.48E+02	5.60E+04	2.65E+07	1.58E+07
Zn	6.37E+01	4.77E+01	2.46E+01	5.91E+06	1.40E+06
Sn	1.31E+01	6.90E+00	2.98E+01		

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

Table 75: LCIA characterization factors for emissions to water: HTP, FAETP, TETP, AET, TET

Name	**HTP**	**FAETP**	**TETP**	**TET**	**AET**
Indicator	Human Toxicity Potential	Freshwater Aquatic Ecotoxicity Potential	Terrestrial Ecotoxicity Potential	Terrestrial Ecotoxicity	Aquatic Ecotoxicity
Source	[1]	[1]	[1]	[2]	[2]
Unit	kg 1,4-dichlorobenzene-eq/kg			kg of triethylene-glycol-eq/kg	
Sb	5.14E+03	1.97E+01	1.66E-20	1.22E-09	2.10E+06
As	9.51E+02	2.07E+02	1.04E-17		3.88E+05
Ba	6.30E+02	2.28E+02	5.08E-19	1.53E-10	8.05E+04
Be	1.40E+04	9.13E+04	3.30E-16		
Pb	1.23E+01	9.62E+00	4.77E-22		2.64E+05
Cd	2.29E+01	1.52E+03	1.42E-20	1.52E-09	2.92E+06
Cr^{III}	2.05E+00	6.91E+00	2.27E-19		
Cr^{VI}	3.42E+00	2.77E+01	2.27E-19		
Cr^{III+VI}	2.07E+00	7.12E+00	2.27E-19		4.53E+05
Co	9.67E+01	3.41E+03	2.69E-18		
Cu	1.34E+00	1.16E+03	4.06E-21		2.06E+07
Mo	5.51E+03	4.76E+02	2.31E-18		
Ni	3.31E+02	3.24E+03	1.03E-18		1.27E+06
Hg	1.43E+03	1.72E+03	9.30E+02	6.93E-08	1.58E+07
Se	5.60E+04	2.92E+03	1.55E-17		3.40E+06
V	3.16E+03	8.97E+03	1.02E-17		
Zn	5.84E-01	9.17E+01	2.53E-21		1.40E+06
Sn	1.73E-02	1.02E+01	7.86E-22		
Fluoride	3.42E+03	1.81E+01	4.28E-05		
Fluorine	3.42E+03	1.81E+01	4.28E-05		
Ammonium				2.18E-02	4.98E+02
NH_3-N				2.80E-02	6.40E+02
NH_4-N				2.80E-02	6.40E+02
Benzo(a)pyrene	2.80E+05	2.52E+05	2.53E-03	2.27E-01	1.71E+06
PAH w/o B(a)P	2.80E+05	2.75E+04	2.12E-03		
Phenols	4.92E-02	2.37E+02	2.49E-06		

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

Table 76: LCIA characterization factors for emissions to water: EP, AEU

Name	**EP**	**AEU**
Indicator	Eutrophication potential	Aquatic eutrophication of inland waters
Source	[1]	[2]
Unit	kg PO_4-eq/kg	kg NO_3-eq/kg
COD	0.022	
TOC	0.044*	
NH_3	0.35	2.45
NH_4	0.33	2.31
NH_3-N	0.42	2.97
NH_4-N	0.42	2.97
NO_3	0.1	0.67
NO_3-N	0.443	2.97
N_{org}	0.42	2.97
HNO_3	0.1	0.67
P species as P	3.06	30.75
PO_4	1.0	10.03
PO_4-P	3.06	30.75
Phosphates (as P_2O_5)	1.34	13.44

** assumption: TOC/COD = 0.5 for treated effluent*
[1]: Guinée et al., 2002
[2]: Hauschild and Potting, 2003 (site-dependent factors for Germany)

Table 77: LCIA characterization factors for extraction of resources: ADP

Name	**ADP**
Indicator	Abiotic Depletion Potential
Source	[1]
Unit	kg Sb-eq/kg
Lignite	6.71E-03
Natural gas	1.87E-02
Crude oil	2.01E-02
Coal	1.34E-02
Hard coal	1.34E-02
Uranium	2.87E-03
Lead	6.77E-04
Iron	4.80E-08
Ferromanganese	6.20E-06
Zinc	3.95E-05
Bauxite	2.10E-09
Sulphur	3.58E-04
Raw phosphate	9.29E-06
Nickel ore, sulphured	5.38E-06
Nickel ore, lateritic	1.08E-06
Raw potassium	3.76E-09
Copper ore (0,99% Cu)	2.20E-05
Chromium ore	2.58E-04
Chromium	8.58E-04

[1]: Guinée et al., 2002 and updated factors from CML method (version 2.7, April 2004, http://www.leidenuniv.nl/cml/ssp/index.html)

12.3 Normalization data for Germany 2004

The following data is used to generate normalization data for the present LCA study which corresponds to the scope of this study. The normalization scores are calculated for Germany as the reference area and 2004 as the reference year. The latest available emission data is used to calculate the normalization scores. They are expressed in relation to a single inhabitant by dividing the total indicator score for Germany by its population (82.532.000 inhabitants in 2004).

Cumulated energy demand

Table 78: Normalization data for cumulated energy demand

	Annual consumption in Germany 2004	CED per inhabitant
	[Petajoule/a]	[MJ/(pe*a)]
CED fossil	12081	146380
CED nuclear	1823	22088
CED renewable	164	1987
CED miscellaneous	370	4483
CED (fossil + nuclear)		**168468**

Source: BMWi, 2005

Climate change

Some substances are neglected (i.e. fluorinated hydrocarbons, chlorofluorocarbons etc), because no emission data for single substances is available. Additionally, these gases are not included in the inventory for the majority of the processes. It is estimated that they contribute less than 2% to the GWP in Germany (UBA, 2007b). Gases with indirect effects on climate change (CO, SO_2, NO_x etc) are neglected due to high uncertainty of their effect factors.

Table 79: Normalization data for global warming potential

Substance	Annual emissions in Germany 2002		Characterization factor	GWP_{100a} per inhabitant
	[1000 Mg/a]	[kg/(pe*a)]	[kg CO_2-eq/kg]	[kg CO_2-eq/a]
CO_2	864117	10470	1	10470
CH_4	3878	47	23	1081
N_2O	180	2.2	296	651
Global warming potential (100a)				**12202**

Source: UBA, 2003

Acidification

Some substances are neglected (i.e. HCl, H_2S) due to lacking emission data. Emission data for HF is estimated as 70% of emission level in 1991 (124000 Mg/a, from IFU and IFEU, 2005).

Table 80: Normalization data for acidification potential

Substance	Annual emissions in Germany 2002		Characterization factor	AP per inhabitant
	[1000 Mg/a]	[kg/(pe*a)]	[kg SO_2-eq/kg]	[kg SO_2-eq/a]
NH_3	607000	7.4	0.77	5.7
HF	86800*	1.1	1.1	1.2
SO_2	608000	7.4	0.6	4.4
NO_x (as NO_2)	1479000	17.9	0.19	3.4
Acidification potential				**14.6**

** estimated with 70% of 1991 emission levels (124000 Mg/a from IFU and IFEU, 2005)*
Source: UBA, 2003

Eutrophication

For COD emissions, emission data is estimated with 120 g COD/(pe*d) (ATV, 2000) and an average elimination of 90% in wastewater plants (DWA, 2005). Normalization scores for eutrophication potential (CML method) are calculated in Table 81, and for terrestrial eutrophication and aquatic eutrophication of inland waters (EDIP method) in Table 82.

Table 81: Normalization data for eutrophication potential

Substance	Annual emissions in Germany 2002		Characterization factor	EP per inhabitant
	[1000 Mg/a]	[kg/(pe*a)]	[kg PO_4-eq/kg]	[kg PO_4-eq/a]
Emissions to air				
NH_3	607000	7.4	0.11	0.81
NO_x	1479000	17.9	0.05	0.9
Emissions to water				
COD	360500*	4.4	0.022	0.1
P species as P	33164	0.4	3.06	1.23
N species as N	687960	8.3	0.42	3.5
Eutrophication potential				**6.5**

**COD emissions are estimated from total COD load per person and day (120 g/(pe*d), ATV, 2000) and mean elimination ratio in wastewater treatment (90%,DWA, 2005)*
Source: UBA, 2003

Table 82: Normalization data for terrestrial eutrophication and aquatic eutrophication of inland waters

Substance	Annual emissions in Germany 2002	Characterization factor		Indicator per inhabitant	
		TEU	AEI	TEU	AEI
	[kg/(pe*a)]	[m² UES per kg]	[kg NO_3-eq per kg]	[m² UES/ (pe*a)]	[kg NO_3-eq/ (pe*a)]
Emissions to air					
NH_3	7.4	80		588	
NO_x	17.9	23.6		423	
Emissions to water					
P species as P	0.4		30.75		12.4
N species as N	8.3		2.97		24.7
Terrestrial eutrophication				**1011**	
Aquatic eutrophication of inland waters					**37.1**

TEU: terrestrial eutrophication
AEI: aquatic eutrophication of inland waters
UES: unprotected ecosystem
Source: UBA, 2003

Human toxicity

Data for calculating normalization scores for human toxicity potential is presented in Table 83 and Table 84. For characterization factors of all relevant substance flows, the reader is referred to annex 12.2. The calculation includes only substances where emission data is available.

Ecotoxicity

Emission data used to calculate normalization scores for ecotoxicity is presented in Table 83 and Table 84. Normalization scores are calculated for terrestrial and freshwater aquatic ecotoxicity potential (indicators from CML method) and terrestrial and aquatic ecotoxicity (IMPACT method). For characterization factors of all relevant substance flows, the reader is referred to annex 12.2. The calculation includes only substances where emission data is available.

Table 83: Normalization data for human and ecotoxicity

Substance	Annual emission in Germany		Source	Indicator				
	[Mg/a]	[g/(pe*a]		HTP	FAETP	TETP	AET	TET
Emissions to 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		
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		
Cr^{VI}	1.15	0.01	2)	3.94E+09	8.84E+03	3.49E+06		
Cr^{III+VI}	115	1.39	1)				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		
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		
PCDD, PCDF	3.09E-4	4.00E-6	1)	5.98E+08	6.57E+05	3.71E+03	1.22E+05	8.00E+01
NH_3	607000	7354	4)	6.07E+07				
HF	86800	1051	5)	2.47E+11	4.03E+08	2.56E+05		1.55E+10
SO_2	608000	7367	4)	5.84E+07				
NO_x (as NO_2)	1479000	17920	4)	1.77E+09				
Dust (PM10)	247000	2993	4)	2.03E+08				

Sources:
1) UBA, 2003 (data from 1995)
2) assumption: 99% Cr^{III} and 1% Cr^{VI}
3) UBA, 2000 (data from 1993/1994)
4) UBA, 2003
5) assumption with 70% of 1991 emission levels (124000 Mg/a from IFU and IFEU, 2005)

Table 84: Normalization data for human and ecotoxicity (continued)

Substance	Annual emission in Germany		Source	Indicator				
	[Mg/a]	[kg/(pe*a]		HTP	FAETP	TETP	AET	TET
Emissions to 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	7.59E+10	
Cd	9.9	0.12	1)	4.53E+08	3.43E+07	3.53E+06	2.89E+10	
Cr^{III+VI}	261.2	3.17	1)	7.36E+07	2.19E+05	3.45E+08	1.18E+11	
Cu	594.6	7.21	1)	3.94E+09	8.84E+03	3.49E+06	1.22E+13	
Ni	324	3.93	1)	4.01E+09	2.28E+05	3.49E+08	4.11E+11	
Hg	4.2	0.05	1)	2.10E+08	7.67E+06	1.30E+06	6.70E+10	
Zn	2931	35.51	1)	3.39E+08	1.75E+07	5.52E+05	4.10E+12	
Emissions to soil								
Pb	1464	17.74	2)	1.86E+08	9.82E+06	8.78E+08	3.86E+11	1.10E+12
Cd	84	1.02	2)	1.19E+09	1.37E+07	1.34E+06	2.44E+11	4.43E+11
Cr^{III+VI}	561	6.80	3)	3.46E+09	1.24E+07	2.72E+06	2.52E+11	1.26E+12
Cu	4577	55.46	2)	4.72E+07	8.04E+06	5.41E+06	9.34E+13	3.17E+13
Ni	476	5.77	3)	7.87E+09	1.21E+06	3.32E+03	6.00E+11	1.57E+12
Hg	3.06	0.04	4)	8.15E+10	3.59E+03	6.68E+02	4.83E+10	8.11E+10
Zn	21237	257.32	2)	2.19E+11	6.57E+07	3.90E+05	2.97E+13	1.26E+14
TOTAL per inhabitant (Emissions to air, water, and soil)				**7266**	**88.9**	**70.1**	**1722063**	**1969880**

Sources:
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

12.4 LCA model for wastewater treatment in activated sludge plant

This part describes in detail the LCA model for conventional wastewater treatment in an activated sludge plant with extended nutrient removal. After basic considerations for LCA models of wastewater treatment, a detailed description of the model explains the allocation of elemental flows and energy demand.

12.4.1 Basic considerations for the LCA model

Concept

This LCA 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 advanced nutrient removal, including oxidation of , nitrification, optional denitrification and chemical P elimination, considering typical parameters of German wastewater treatment plants (WWTP). Sludge treatment comprises of simultaneous aerobic stabilisation or anaerobic digestion, followed by dewatering of residual sludge. If the excess sludge is digested, biogas from the digestion process is combusted in a combined heat and power plant (CHP plant) to provide electrical and thermal energy for the operation of the WWTP.

The input substance flows are allocated to the output flows of discharged water, sludge and air specifically for each elemental flow. In addition, material and energy demands for WWTP operation are calculated from input flows and operating conditions.

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.

12.4.2 Model description

12.4.2.1 Wastewater composition

The input wastewater is characterized in terms of 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 to NH_4-N, organic nitrogen, particulate nitrogen). Therefore, exemplary parameters describe the different fractions of the elemental flows (Table 85). While particulate species can be separated physically by sedimentation, dissolved species have to be treated biologically (activated sludge) or chemically (precipitation).

Table 85: Assumed composition of influent wastewater

Species		Proportion [%]
Nitrogen	NH_4-N	54
	NO_3-N	0
	N organic	35
	N particulate	11
Phosphorus	PO_4-P	83
	P particulate	17
Sulphur	SO_4-S	96
	S particulate	4

12.4.2.2 Wastewater treatment

The present WWTP model describes the processes of mechanical, biological and chemical (or primary, secondary and tertiary) wastewater treatment, including sludge treatment with thickening, stabilisation and dewatering. The removal efficiency for the most important wastewater constituents (COD, nitrogen, and phosphorus, all of which are regulated in legal standards for municipal wastewater treatment (ATV, 2000)) is adjustable via parameters (Table 86). The model calculates the specific elemental allocation of wastewater components to the different output flows (discharged water, air, stabilized sludge, biogas) and the associated demand for chemicals or energy. 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. In the following, the most important elements are discussed in detail.

Table 86: Parameters of LCA model for conventional WWTP

Parameter	Description		
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 gas	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%)	90	%
PE*	Efficiency of dissolved phosphorus removal	95	%
PFE*	Proportion of phosphorus removal via precipitation (rest: biological P elimination)	60	%
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	days
YC	Yield coefficient (g C biomass per g C substrate)	0.67	--

** effective parameter values in this study depend on scenario*

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. 94% for COD and 98% for biological oxygen demand (BOD) (DWA, 2005). The particulate fraction of COD is separated by sedimentation in primary treatment, so no aeration energy is required for its removal in the activated sludge process. In case of sludge digestion, sludge from primary treatment increases the production of biogas.

The dissolved COD is partially transformed into biomass (assimilation) and partially into CO_2 and water (dissimilation) during the activated sludge process. 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 NH_4-N, organic-bound nitrogen, and particulate nitrogen. The removal of total nitrogen amounts to 70 – 90% in average WWTPs if specific denitrification is employed. In average, 81% of total nitrogen load are eliminated in German WWTPs (DWA, 2005). 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 the effluent is assumed to 70% NO_3-N, 20% N org., and 10% NH_4-N. The low fraction of NH_4-N in the effluent reflects an almost complete nitrification process (> 90% oxidation of $NH_{4,in}$).

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 from the

tanks. In this study, 0.6 % of denitrified N is transformed into N_2O (Wicht, 1996) and 0.6 % of influent NH_4 into gaseous NH_3 (adopted from Bardtke et al., 1994).
There are different concepts for a nitrogen balance of wastewater treatment. The nitrified NH_4-N can be estimated as follows (Scheer, 1998):

$$(NH_4\text{-}N)_{NIT} = TKN_{influent} - N_{excess\ sludge} - NH_4\text{-}N_{effluent} - N\ org_{effluent}$$

Denitrified nitrogen is calculated by the following equation:

$$(NO_3\text{-}N)_{DEN} = (NH_4\text{-}N)_{NIT} + NO_3\text{-}N_{influent} - NO_3\text{-}N_{effluent}$$

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

$$N_{excess\ sludge} = 0.25 * TKN$$

Phosphorus

Phosphorus content of wastewater consists of dissolved and particulate phosphorus species. It is assumed that particulate phosphorus is completely separated by sedimentation during primary treatment. 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).

EBPR requires a specific process engineering and favourable wastewater composition (Scheer and Seyfried, 1996). In this study, biological P elimination is attributed only to usual incorporation in biomass, without intended EBPR mechanisms. Thus, 40% of dissolved phosphorus is eliminated without chemical P elimination. In case of extended nutrient elimination, P removal (> 95%) is reached by the addition of ferric salts ($FeCl_3$ and $FeSO_4$).

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. Sulfate from addition of flocculants ($FeSO_4$) is added to the effluent.

Heavy metals

Inorganic trace substances like heavy metals are removed in varying proportions from the wastewater and transferred into the sludge. The degree of removal is determined by their speciation and different adsorption properties to the wastewater sludge. The remaining heavy metals are discharged with the effluent. Transfer coefficients of heavy metals are compiled from various literature sources (Table 87).

Table 87: Transfer of heavy metals to sewage sludge in activated sludge plant

Element	Transfer to sewage sludge in % of influent load		
	This study	Literature	Source
Lead	80	88 / 62	1 / 2
Cadmium	70	73 / 50	1 / 3
Chrome	80	85 / 50	1 / 4
Copper	85	88 / 46 – 93	1 / 5
Nickel	60	63 / 40	1 / 4
Mercury	80	79 / 91 / 70	1 / 2 / 3
Zinc	75	79 / 60 – 70	1 / 3

1) Fuchs et al., 2002 (mean values of literature review for activated sludge plants with P elimination)
2) Raach et al., 1999
3) Koppe and Stozek, 1999
4) Zimmermann et al., 1996
5) Overath et al., 1997

Inorganic salts

Most inorganic salts are highly soluble in water. Hence, it is estimated that they remain in the discharged water except for the fractions in sludge water content. Chloride from addition of flocculants ($FeCl_3$) 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, which are typical wastewater components of anthropogenic nature, e.g. chlorinated chemicals. 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 mechanical treatment are neglected in this study. Usually, the screenings are disposed of in a landfill after washing, dewatering, and compaction. The sand or grit is washed to remove residual organic material, and it can be reused e.g. for road construction after classification.

Flocculation chemicals

Chemical P elimination in wastewater treatment can be achieved by the addition of ferric salts (= flocculation). The two salts which are usually used for flocculation are ferric (III) chloride and ferrous (II) sulfate. Both chemicals have basically the same effect for P elimination, although the use of $FeSO_4$ consumes less alkalinity. In this study, both flocculants are used in equal amounts, i.e. 50% of the required molar mass of iron is delivered as $FeCl_3$ (ferric chloride, 40% solution) and 50% as $FeSO_4$ (ferrous sulfate 7-hydrate, 96% solution). Potential contamination of flocculants with heavy metals is neglected here.

Decisive for the applied flocculant 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 which represents a common dosage for wastewater treatment (Scheer, 1998).

12.4.3 Sludge treatment

Sewage sludge contains a considerable fraction of degradable organic substances, which can lead to the formation of malodorous or toxic gases during storage and disposal of the sludge. Hence, the sludge has to be stabilised prior to its disposal to prevent uncontrolled degradation processes. It can either be stabilised by continuous aeration of the sludge (= simultaneous aerobic stabilisation) or by digestion (= anaerobic stabilisation). Both types of stabilisation are included within the reference scenarios.

The LCA model for conventional wastewater treatment developed at TU Berlin was initially designed to describe the operation of large wastewater treatment plants. These plants are typically equipped with a digester for anaerobic sludge stabilisation. Thus, the process inventory for sludge digestion and sewage gas usage is readily available and is described below.

For reasons of simplicity, the present model is also used to describe the process of simultaneous aerobic stabilisation. Minor changes in the calculation of energy demand and emissions are necessary to adopt the process model for sludge digestion to aerobic sludge stabilisation with adequate accuracy (see below).

12.4.3.1 Sludge digestion and sewage gas usage

Raw sludge

Raw sludge is composed of the organic and inorganic particulate substances from sedimentation (primary sludge), the generated biomass of the activated sludge process (excess sludge), and the remaining inert solids from endogenous decomposition of biomass (ATV, 2000). A high sludge age leads to more respiration of carbon into CO_2 and less organic matter in the excess sludge. In case of chemical P elimination, the excess sludge additionally contains precipitated iron from flocculation with ferric salts. The elemental composition of the raw sludge is calculated as follows:

- The contents of carbon, nitrogen, phosphorus, sulphur, iron, heavy metals and salts are calculated from input loads with transfer coefficients
- Organic matter is mainly composed of carbon, oxygen and hydrogen. The contributory fraction of oxygen and hydrogen to the organic matter is calculated with proportional factors related to carbon (oxygen: 70% of carbon, hydrogen: 15% of carbon).
- In case of chemical P elimination, the equivalent oxygen and hydrogen content of the precipitated iron is calculated by assuming iron in the form of $Fe(OH)_3$.
- The mass of the inert fraction in the raw sludge is estimated to 45% of its organic content.
- For the conversion of chemical oxygen demand in excess sludge into organic matter, the factor 1.45 g COD/g organic dry matter is applied.

Sludge thickening

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

Digestion

This process model describes a mesophilic sludge digestion process (as for example in Gujer, 1999). The thickened raw sludge is digested in a mesophilic reactor (33 – 37°C). Within an average retention time of 15 – 30 days, degradable organic matter is converted into biogas. The process needs a thermal energy input of 130-180 MJ/m³ raw sludge to maintain the required operating temperature.

Around 40 – 50% of the organic matter content of the raw sludge is converted to biogas, generating ca. 0.9 m³ biogas per kg decomposed organic matter. The biogas (density: 1.15 kg/m³) 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 theoretical content of CH_4,

CO_2, NH_3, and H_2S of the biogas can be calculated by the disproportioning equation (Tidden, 2003):

$$C_aH_bO_cN_dS_e + (4a - b - 2c + 3d + 2e) \cdot H_2O \longrightarrow$$
$$\tfrac{1}{8} \cdot (4a + b - 2c - 3d - 2e) \cdot CH_4 + \tfrac{1}{8} \cdot (4a - b + 2c + 3d + 2e) \cdot CO_2 + d \cdot NH_3 + e \cdot H_2S$$

The organic matter of raw sewage sludge is usually 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 with the following transfer coefficients:

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

These transfer coefficients are used for the carbon conversion in the digestion process. The decomposition of proteins leads to the production of NH_4, which is recycled to the influent with the sludge liquor. 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 recycled NH_4 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). The dosage of polyacrylamide is set to 7 g/g dry matter. A stabilised sludge with good properties can be dewatered to a final dry matter content of 40%.

Sewage gas usage in CHP plant

The sewage gas is combusted in a combined heat and power plant (CHP plant) to generate electrical and thermal energy. A potentially required gas conditioning (e.g. drying) prior to the combustion process is neglected. A part of the generated sewage gas has to be flared in case of system malfunction or storage overflow. It is assumed that 5% of the total sewage gas volume is flared (Ronchetti et al., 2002), generating emissions which are comparable to the combustion in the CHP plant. A small proportion of the sewage gas (0.75%) is lost by accidental leakage (Ronchetti et al., 2002), causing respective emissions of methane.

The CHP plant is equipped with a spark-ignition engine ("otto engine") in lean combustion mode with high excess air, so that legal air emission standards can be met easily. CHP parameters and emission factors are compiled from an LCA study (Ronchetti et al., 2002) and the Umberto® database (IFU and IFEU, 2005)

Table 88: Parameters and emission factors for CHP plant

	This study	Biogas-CHP	Natural gas-CHP
Source		Ronchetti et al., 2002	Umberto® database (IFU and IFEU, 2005)
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
Emissions in mg/MJ			
CH_4, combustion	2,5	2,48	3,78
CO_2	*	81.308	55.151
NO_x (as NO_2)	38	37,85	62,98
N_2O	1,6	--	1,57
CO	51	50,93	51,17
SO_x (as SO_2)	30	29,91	0,43
NMVOC	2,5	2,48	4,72
Dust	1,6	--	1,57

** depending on input (CO_2 + CH_4) minus CO*

12.4.3.2 Aerobic stabilisation of sludge

For a simplified model of aerobic stabilisation of wastewater sludge, the LCA model of anaerobic sludge treatment is slightly modified. Aerobic stabilisation of wastewater sludge is done simultaneously to the normal activated sludge process just by extended aeration of the sludge (Gujer, 1999). This process is typically operated at smaller wastewater plants. For adopting the LCA model to describe simultaneous aerobic sludge stabilisation, the following modifications are implemented:

- Sludge age is increased to 25 days (ATV, 2000), leading to a decrease in excess sludge production
- Generated gas in stabilisation is only CO_2 (= transfer of $C_{raw\text{-}sludge,\ decomposed}$ into 93% CO_2-C and 7% HCO_3-C)
- Energy demand for aeration of wastewater and for recirculation and mixing of aeration tank is increased by 50% (Müller et al., 1999)
- No energy demand for digester operation
- No thermal energy required for sludge heating
- Thickening, dewatering, and sludge transport is similar to sludge digestion

12.4.4 Energy demand

The energy demand of wastewater treatment plants has been extensively studied in literature (e.g. LfU, 1998; Müller et al., 1999) to identify important electric consumers and optimize the energy balance of wastewater treatment. Many factors influence the actual energy demand of a certain WWTP: Input-related parameters (influent volume, COD and N load), the process layout and the plant dimension are important. Average energy data from 1097 municipal wastewater treatment plants in Germany (Table 20) show a wide range in energy demand related to influent volume, degraded COD, and population equivalents.

Table 89: Energy demand of 1097 municipal WWTP in Germany

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

Source: LfU, 1998

For this LCA model, specific energy coefficients for the different sub-processes of wastewater treatment are calculated (Table 90). The calculation of the coefficients is based on index figures of energy demand and specific data of wastewater volume, load and removal efficiency (Müller et al., 1999; LfU, 1998). With these coefficients, the LCA model calculates the effective energy demand for wastewater treatment in relation to influent quality and quantity and preset removal efficiencies.

Table 90: Allocation of energy demand in wastewater treatment plants

	Electric 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 for carbon degradation	0.55*	kWh/kg $COD_{respirated}$ (→ Table 92)
Aeration for nitrification	2.34	kWh/kg $N_{nitrified}$
Benefit from denitrification	-1.58	kWh/kg $N_{denitrified}$
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	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**		150 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	kWh/m³ wastewater 38 kJ/m³ wastewater

Sources: LfU, 1998; Müller et al., 1999
** for WWTP with aerobic stabilisation, these values are increased by 50%*
*** thermal energy*

Oxygen demand and aeration energy for carbon degradation and nitrification

The oxygen demand for carbon degradation and endogenous respiration, nitrification and denitrification (Table 91) is estimated according to common design rules for wastewater treatment plants (ATV, 2000). 62.5% of the oxygen demand for nitrification is reclaimed for microbial processes via the use of NO_3-oxygen during denitrification.

Table 91: Specific oxygen demand of biological wastewater treatment

Process		Value or calculation	
Carbon degradation + 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	(= oxygen benefit from NO_3)

Source: ATV, 2000

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 calculated above. The specific energy demand for effective oxygen transfer (0.55 kWh/kg $O_{2,\ effective}$) in fine-bubble aeration is calculated with common design parameters for activated sludge plants (Imhoff, 1990). The energy demand for oxygen transfer is used to calculate the specific energy demand for aeration for carbon degradation, nitrification, and denitrification (Table 92).

Table 92: Calculation of specific energy demand for aeration

Parameter		Value
Energy demand for fine-bubble aeration	kWh/(kg O_2*m)	0.02*
Injection depth	m	3*
Oxygen utilization (3 m injection depth)	%	11*
Specific energy demand for oxygen transfer	kWh/kg $O_{2,effective}$	0.55
Specific energy demand for aeration		
Carbon degradation + endogenous respiration	kWh/kg $COD_{respirated}$	0.55
Nitrification	kWh/kg $N_{nitrified}$	2.34
Denitrification	kWh/kg $N_{denitrified}$	- 1.58

**Source: Imhoff, 1990*

12.5 Agricultural tractor

For the calculation of the respective emissions and fuel demand of the tractor, a three step procedure is necessary:

1. calculation of field size (in ha) to which the specific fertilizer is applied
2. definition of working time per ha and engine load levels during fertilizer application
3. calculation of total working time and inserting all information into a given data set for tractor operation

Calculation of the area of farmland for fertilizer application
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 year
K_2O: 160 kg/ha, one dose per year
N: 200 kg/ha, split into four doses per year

Table 93 lists an exemplary distribution of the required nutrients on different types of fertilizer, taking into account the approximate nutrient content. It is assumed that farmers will apply a mixture of all available types of fertilizer to provide suitable amounts of nutrients to their crops.

Table 93: Exemplary distribution of the nutrient amounts of manure, mineral and organic fertilizers for the cultivation of winter wheat

Nutrient	Demand	Manure	Mineral fertilizer	Secondary organic fertilizers
	[kg/ha*a]	[kg/ha*a]	[kg/ha*a]	[kg/ha*a]
P_2O_5	90	63	0	27
K_2O	160	95	34	31
N	200	90	0	110

Of all nutrients, nitrogen is the most important during the growing period of the crops. Consequently, the field size to which the fertilizers are applied is calculated based on the nitrogen demand. The source separated urine contains around 6.7 kg/m³ N, which is equivalent to 11272 kg plant-available N per year in total for the settlement (N losses are already subtracted).

Assuming the fertilizer management of Table 93, the collected urine can be applied to an area of 102 ha (Table 94). The application area for the other fertilizers is calculated accordingly (110 kg N/(ha*a)). For the different composts, the application rate is limited by a maximum applicable amount of 25 t compost per ha and year. Due to its low content of plant-available nitrogen, the N input on the respective field area via compost is relatively low. For the application of sewage sludge, a maximum amount of 1.66 t/(ha*a) dry matter is regulated by law (AbfKlärV, 1992).

Table 94: Calculation of field area for fertilizer application in each scenario

Scenarios	Fertilizer	Plant-available nitrogen	Nitrogen dosage	Field area
		[kg/a]	[kg N/(ha*a)]	[ha]
Reference (R_{min} + R)	Mineral	16132	110	147
	Compost (biowaste)	143	20.4*	7
Reference (R_{agri})	Mineral	14121	110	128
	Compost (biowaste)	143	20.4*	7
	Sewage sludge	2011	39.4**	51
Faeces composting + urine separation (SC)	Mineral	4683	110	43
	Urine	11272	110	102
	Compost (faeces +biow.)	320	26.7*	12
Faeces digestion + urine separation (SV)	Mineral	3380	110	31
	Urine	11272	110	102
	Digester residual (stabilised)	1630	110	15
Faeces and urine digestion (V)	Digester residual (liquid)	16280	110	148

** limited by a maximum applicable amount for compost: 25 t/(ha*a)*
*** limited by a maximum applicable amount for sewage sludge: 1.66 t/(ha*a) dry matter*

Specific working time per ha and engine load levels of agricultural tractor for each fertilizer

Solid mineral fertilizers are applied with various methods (e.g. centrifugal spreader). Liquid fertilizers are usually applied with splash plates, nozzles, trailing hoses, or liquid injection. To minimize nitrogen losses during urine application, trailing hoses or injection systems should be applied. In addition, the liquid fertilizer can be incorporated into the ground by harrowing. A dilution of urine with service water (3:2) prior to the application further reduces N losses. Composts are loaded on trailers with a crane and applied via manure spreaders.

Table 95 lists the estimated working time and the distribution of engine load levels for the application of mineral fertilizer, urine or liquid digester residual, and composts. Data is compiled from charts of a detailed study on agricultural machinery (Rinaldi et al., 2005).

Table 95: Working time and distribution of engine load levels during the application of different fertilizers

Type of fertilizer		Mineral fertilizer	Urine#, sewage sludge or liquid digester residual	Compost	
Nominal engine power	[kw]	50	50	50	50
Remarks		Spreader width: 15 m	Pressure drum: 6.5 m³	Loading	Spreading
Working time*	[h/ha]	0.7	1.6	1.5	2.3
Applied amount		--	30 m³/ha	25 t/ha	
Engine load levels					
Heavy load	[%]	10	10	0	20
Medium load	[%]	60	50	0	30
Easy load	[%]	0	0	95	10
Road	[%]	20	30	0	20
Engine idle	[%]	10	10	5	20
Calculated fuel demand**	[L/ha]	3.8	8.5	16.7	

Source: Rinaldi et al., 2005
** incl.all working steps, road traffic etc; field size: 2 ha, distance farm to field: 1000 m*
*** calculated with data set "agricultural tractor" (IFU and IFEU, 2004)*
 # diluted with service water (3:2)

Total working time for fertilizer application

In Table 96, the total working time for the application of the respective amounts of fertilizer is listed for each scenario. The calculated times are based on the assumption of different fertilizing regimes. For compost, a single application per year is assumed to fulfil its function as a long-term fertilizer and soil conditioner. Urine, sewage sludge and liquid fertilizer are applied in two doses per year due to their relatively high nutrient content. Mineral fertilizer is applied in three doses per year (two for nitrogen, one for phosphorus and potassium).

The respective values are fed into the data set "agricultural tractor" provided by UMBERTO® (IFU and IFEU, 2004). This data set calculates the energy demand and the emissions from agricultural tractor operation depending on engine power, total working time, and engine load levels (dataset based on Borken et al., 1999).

Table 96: Calculation of total working time during fertilizer application

Scenarios	Fertilizer	Field area	Time per area	Regime*	Total working time
		[ha]	[h/ha]	[doses/a]	[h/a]
Reference (R_{min} + R)	Mineral	147	0.7	3	308
	Compost (biowaste)	7	1.5 + 2.3	1	11 + 16
Reference (R_{agri})	Mineral	128	0.7	3	270
	Compost (biowaste)	7	1.5 + 2.3	1	11 + 16
	Sewage sludge	51	1.6	2	165
Faeces composting + urine separation (SC)	Mineral	43	0.7	3	89
	Urine	102	1.6	2	328
	Compost (faeces +biow.)	12	1.5 + 2.3	1	18 + 28
Faeces digestion + urine separation (SV)	Mineral	31	0.7	3	65
	Urine	102	1.6	2	328
	Digester residual (stabilised)	15	1.5 + 2.3	1	23 + 35
Faeces and urine digestion (V)	Digester residual (liquid)	148	1.6	2	472

** assumed number of doses per year*

12.6 Data for construction of sanitary systems

This part describes in detail the calculation of the material demand for the construction of the infrastructure of all scenarios. Infrastructure can be divided into three main parts: the sanitary installations inside the buildings, the sewer system, and the facilities for treatment and storage. A map of the settlement area in Berlin which is used for the exemplary layout of the sewer system is provided below (Figure 93).

12.6.1 Map of the settlement area

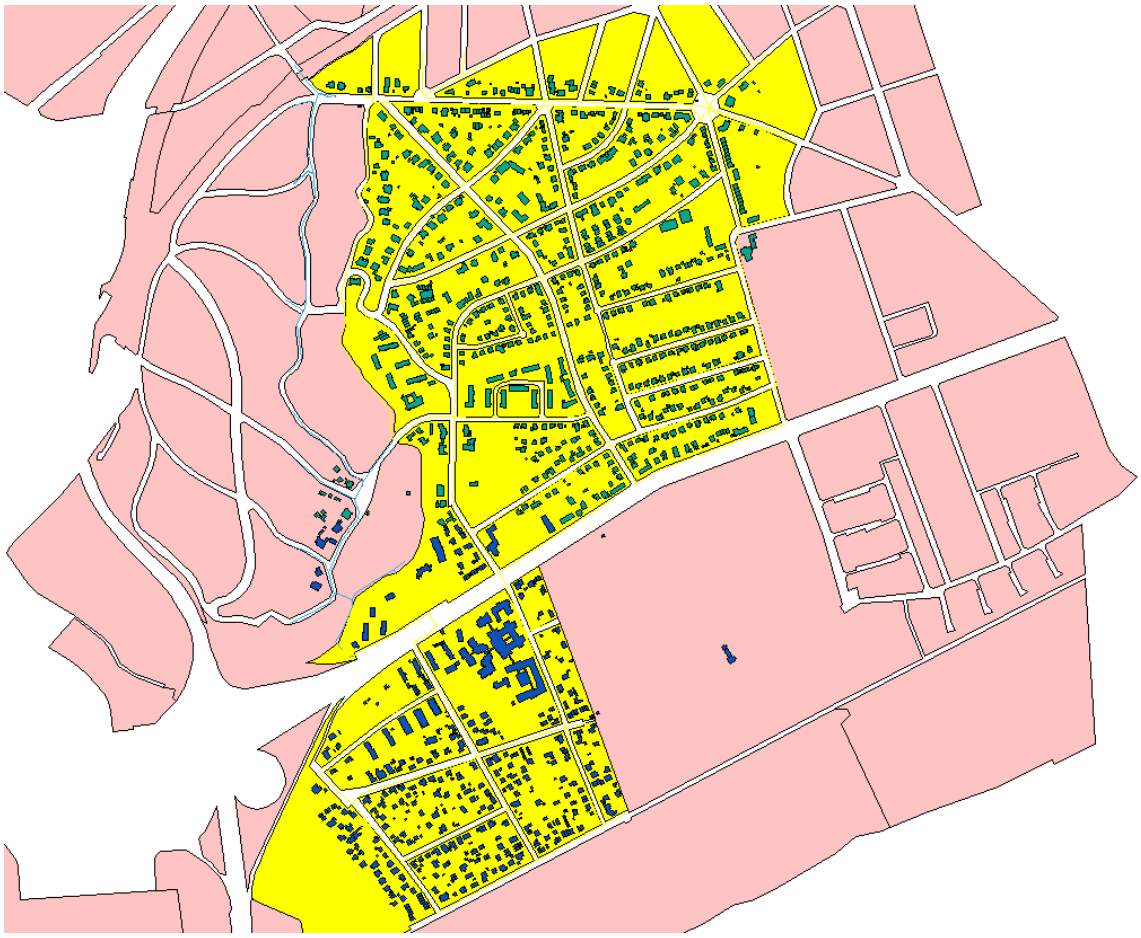

Figure 93: Map of the settlement area (Berlin-Nicolassee, study area is marked in yellow), source: Berliner Wasserbetriebe (www.bwb.de), screenshot from GIS data software ARCVIEW®

12.6.2 Sanitary in-house installations

The sanitary piping of prototype houses (Figure 94) and apartments (Figure 95) is designed according to the relevant German legal norm (DIN EN 12056-2, 2001). Details of the calculation of sanitary in-house-piping for conventional and source-separation systems are presented in Table 97 and Table 98.

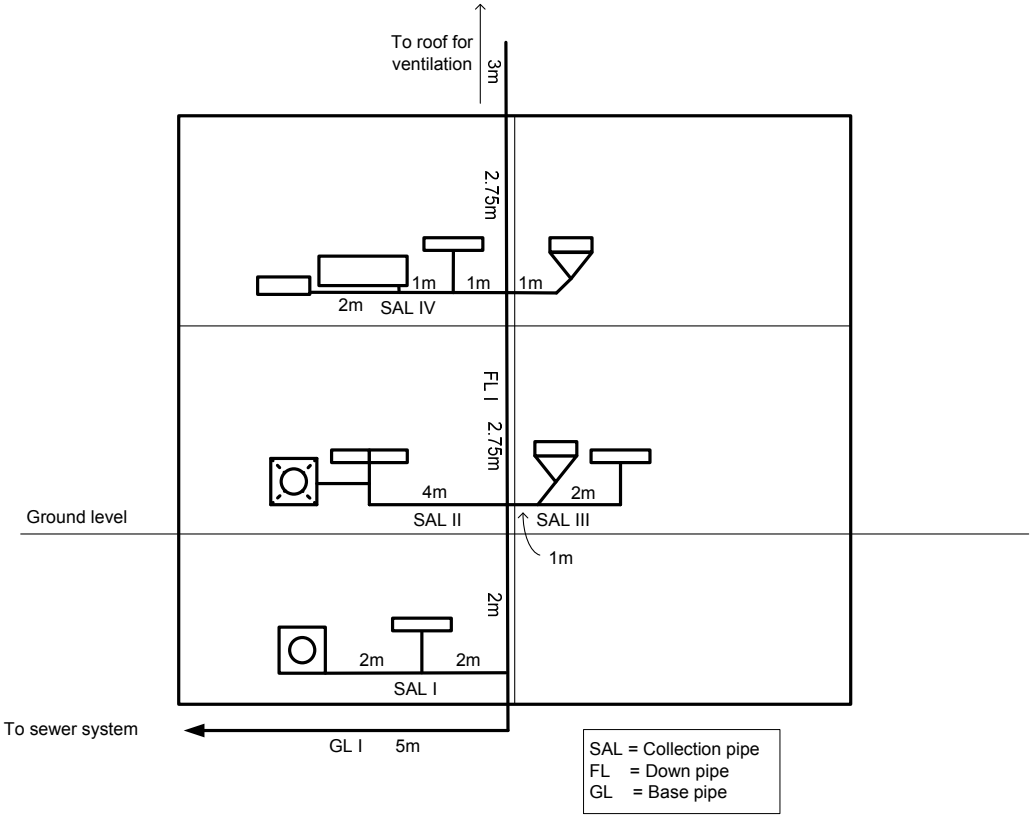

Figure 94: Prototype house for dimensioning of sanitary in-house installations

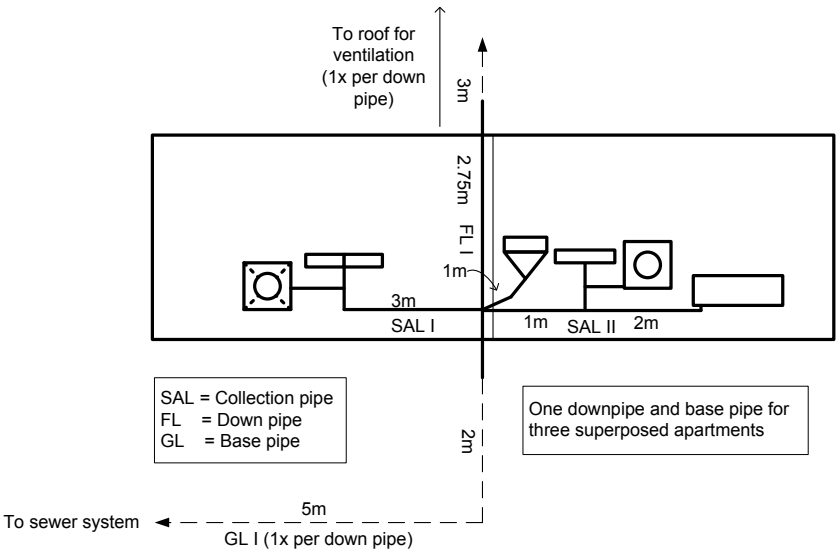

Figure 95: Prototype apartment for dimensioning of sanitary in-house installations

Table 97: Inventory of in-house installations (conventional system)

Building	Line	Pipe material and diameter	Length [m]	Sum [m]	Total [m]	Final length** [m]
House	SAL I + II	PP 50	4 + 4	14	12180	14007
	SAL III + IV	PP 50	2 + 4			
	SAL III + IV	PP 100	1	12.5	10875	12506
	Down pipe	PP 100	10.5			
	Base pipe	PE 150	5	5	4350	4785
Apartment	SAL I + II	Cast iron 50	3 + 3	6	7020	7722
	SAL II	Cast iron 100	1	16.25	6337.5	6971
	Down pipe*	Cast iron 100	13.25			
	Base pipe*	PE 150	5	15	1950	2243

** one down pipe and base pipe for three superimposed apartments*
*** including proportional factors for fittings (+10% for PE/cast iron, +15% for PP)*

Table 98: Inventory of in-house installations (source-separation)

Building	System**	Line	Pipe material and diameter	Length [m]	Sum [m]	Total [m]	Final length*** [m]
House	F	SAL III + IV	PP 100	1 + 1	12.5	10875	12506
		Down pipe	PP 100	10.5			
		Base pipe	PE 150	5	5	4350	4785
	GW	SAL I + II	PP 50	4 + 4	15	13050	15008
		SAL III + IV	PP 50	3 + 4			
		Down pipe	PP 70	6.25	6.25	5437.5	6253
		Base pipe	PP 100	5	5	4350	5003
	U	SAL III + IV	PP 50	1 + 1	2	1740	2001
		Down pipe	PP 70	6.25	6.25	5437.5	6253
		Base pipe	PP 100	5	5	4350	5003
	Vac	Total	PE 50	12	12	10440	11484
	SWS	Total	PE 20	13.25	13.25	11527.5	13257
Apartment	F	SAL II	Cast iron 100	1	5.42	6337.5	6971
		Down pipe*	Cast iron 100	13.25			
		Base pipe*	PE 150	5	1.67	1950	2145
	GW	SAL I + II	Cast iron 50	3 + 3	6	7020	7722
		Down pipe*	Cast iron 70	9	3	3510	3861
		Base pipe*	PP 100	5	1.67	1950	2243
	U	SAL II	PP 50	1	1	1170	1346
		Down pipe*	PP 70	9	3	3510	4037
		Base pipe*	PP 100	5	1.67	1950	2243
	Vac	Total*	PE 50	17	5.67	6630	7293
	SWS	Total	PE 20	5.67	5.67	6630	7625

** one down pipe and base pipe for three superimposed apartments*
*** F: gravity drainage of faeces, GW: greywater, U: urine, Vac: vacuum system, SWS: service water supply (for greywater reuse)*
**** including proportional factors for fittings (+10% for PE/cast iron, +15% for PP)*

12.6.3 Drainage systems

The layout of the drainage systems is designed following common German regulations (ATV, 1999; DIN EN 752, 1997). Figure 96 shows an exemplary map of the conventional sewer system in the northern part of the study area. Resulting pipe lengths for the different systems (Table 99) are used to calculate total material demand with weight factors from manufacturers. Additional components such as house shafts, inspection chambers or holding tanks are included in the inventory through quantity (Table 100) and weight of single components (Table 101).

Figure 96: Exemplary layout of the conventional sewer system in the northern part of the settlement area (Red: sewage pipes)

Table 99: Pipe dimensions, materials, and total lengths for sewer systems

Material	Ø	Weight	Ref	Grey-water	Service water	Faeces		Urine	Fitting factor*
						Gravity	Vacuum		
	[mm]	[kg/m]	[m]	[m]	[m]	[m]	[m]	[m]	[%]
House connections									
Vitr.clay	150	24	10000	0	0	10000	0	0	10
PP	150	1.943	0	0	0	0	0	10000	15
PE	32	0.433	0	0	10000	0	0	0	15
	50	0.457	0	0	0	0	10000	0	20
	150	2.46	0	10000	0	0	0	0	20
Drainage									
Vitr.clay	150	24	11250	0	0	11108	0	0	10
	200	36	1010	0	0	0	0	0	10
	250	51	645	0	0	0	0	0	10
	300	67	370	0	0	0	0	0	10
	400	104	530	0	0	0	0	0	10
PE	50	0.457	0	0	0	7816	0	7050	15
	65	0.727	0	0	0	0	14185	0	15
	100	2.18	0	0	0	0	1364	0	15
	150	2.46	0	13168	0	0	0	0	15
(pressure)	150	4.56	0	0	11108	0	0	0	15
	200	4.82	0	470	0	0	0	0	15
PP	150	1.943	0	0	0	0	0	11108	20

**proportional addition for fittings (PP: including 5% for plug-in connections)*
Sources for weight factors:
Vitrified clay: Euroceramic, 2005
Polypropylene (PP):Ostendorf, 2005
Polyethylene (PE): Simona, 2005

Table 100: Additional components of sewer systems

System	Component	Material	Dimensions		Quantity
				Depth [m]	
Conventional	House shafts	Concrete	DN 1000	2	1000
	Inspection chambers	Concrete	DN 1000	3	342
Faeces with gravity drainage	House shafts	Concrete	DN 1000	2	-*
	Inspection chambers	Concrete	DN 1000	3	-*
	Pumping shafts	Concrete	DN 1000	3	32
Vacuum	Inspection chambers	Concrete	DN 1000	1	67
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

** one house shaft and inspection chamber for brownwater, greywater and service water*

Table 101: Material weight of chambers and shafts

Component	Material	Dimensions		Weight	Source
			Depth [m]	[kg]	
Inspection chambers	Concrete	DN 1000	3	3050	Mall, 2005
	Cast iron			35	estimated
House shafts	Concrete	DN 1000	2	1610	Mall, 2005
	Cast iron			26	estimated
Pumping shafts	Concrete	DN 1000	3	3050	Mall, 2005
	Cast iron			35	estimated
Urine inspection shafts	Polypropylene	DN 400	3	40	Ostendorf, 2005
Manhole covers	Concrete	DN 800		80/75*	Mall, 2005
	Cast iron			40/120*	estimated

** class B125/D400: house shafts are B125, rest is D400*

12.6.4 Treatment facilities

The design of the treatment facilities heavily depends on local conditions and specific process layout. Therefore, the assessment of the material demand for the treatment facilities can only roughly estimate the material demand for a potential sanitation system. In general, the difficult acquisition of appropriate material data for the various installations often leads to the use of qualified estimates. Estimates are based on data of comparable processes, manufacturer product sheets, or other available information. This section provides information about the material demand of:

- conventional wastewater treatment plant
- greywater treatment units (SBR, MBR + service water storage, soil filter)
- urine collection tanks
- solid-liquid separation process for faeces dewatering
- vacuum system
- biogas plant.

The composting plant for biowaste treatment or combined treatment of faeces and biowaste is excluded from this study as well as the ozonation unit for urine treatment.

Conventional wastewater treatment plant

Construction data for the conventional wastewater treatment plant is adopted from a detailed construction inventory of a German activated sludge plant (design dimension: 21000 inhabitant equivalents) (Schneidmadl, 1999). This data is converted to the plant dimension of this study (~ 5000 inhabitant equivalents) by recalculating the material demand in relation to the daily influent volume (m³/d) with a 10% safety margin ($V_{design\ calc} = V_{daily\ wastewater} * 1.1$) (Table 102).

SBR and MBR for greywater treatment

For the construction of greywater treatment plants (SBR and MBR), adequate inventory data is not available. Hence, it is roughly estimated that both processes (conventional activated sludge plant and SBR/MBR for greywater treatment) have a comparable material demand in relation to the treated volume. Both SBR and MBR for greywater treatment are described with the same inventory dataset which is recalculated in relation to the daily influent volume (Table 102).

For MBR plants, the tank volume of the activated sludge tank can typically be reduced by a factor of 3-4 (Pinnekamp and Friedrich, 2006), and no clarifier is required. Consequently, construction data for the MBR plants for greywater treatment is corrected by a factor of 4 for those materials which are relevant for tank construction (concrete, non-alloy steel, and excavation). The possibility of a decentralized layout of

the greywater reuse system (with several small MBR units distributed across the settlement) to minimize the transport distances for greywater drainage and service water supply is not accounted for in this study.

The purified greywater is temporarily stored in holding tanks before the distribution as service water. The volume of the stored water should be large enough to balance variations in the amount of treated greywater and service water demand. On the contrary, a long retention time in the storage tank promotes the risk of bacterial regrowth and associated problems (e.g. odour). Following the recommendations of the German association for the use of service and storm water (Fbr, 2005), the volume of the storage tanks is calculated to provide the average service water demand of one day (= 120 m^3 as maximum daily demand in scenario SC3). The material demand for the storage tanks (10 x 12 m^3) is estimated in analogy to the urine storage tanks: a 12 m^3 tank made of glass fibre reinforced plastic weights 800 kg according to manufacturer data (Mannschott, 2005).

Table 102: Construction data for a conventional wastewater treatment plant and calculated material demand for wastewater or greywater treatment plants in this study

System		Conventional activated sludge plant (Schneidmadl, 1999)	Combined wastewater	SBR greywater		MBR greywater	
				with faeces filtrate	*without faeces filtrate*	*with faeces filtrate*	*without faeces filtrate*
Influent volume	*[m³/d]*	7540	638	572	440	572	440
Influent volume*	*[L/pe*d]*		127.6	114.4	88	114.4	88
Material	*[kg]*						
Concrete		30809720	2606976	2337289	1797915	584322#	449479#
Steel, high-alloy		57390	4856	4354	3349	4354	3349
Steel, non-alloy		498020	42140	37781	29062	9445#	7266#
Polyethylene HD		28820	2439	2186	1682	2186	1682
Limestone		602580	50988	45713	35164	45713	35164
Copper		13310	1126	1010	777	1010	777
Aluminium		3090	261	234	180	234	180
Glas		1130	96	86	66	86	66
Excavation		96382600	8155451	7311783	5624448	1827946#	1406112#

values are calculated in relation to influent volume (m³/d)
** influent volume is calculated as daily volume + 10% safety factor*
 # assumption for MBR plants: tank volume can be reduced by a factor of 4 (Pinnekamp and Friedrich, 2006), relevant for concrete, non-alloy steel, and excavation

Soil filter for greywater treatment

Greywater and faeces filtrate are treated in planted soil filters in scenarios SC2, SV2, and V2. For this treatment, three large soil filters are designated for greywater treatment near the settlement. Each soil filter consists of an upstream sedimentation tank (HRT_{min}~ 3h, volume: 40 m³), a feeding pump, and a planted soil filter with ground sealing (PE foil + PES fleece) and pipes for feeding and drainage (Table 103).

For the dimensioning, the necessary surface area for the soil filters is calculated according to recommendations of ATV (ATV, 1998) and operational experience from pilot sites in Lübeck-Flintenbreite (Oldenburg, 2002) and Stahnsdorf (Peter-Fröhlich et al., 2007). In scenarios without faeces filtrate (SV2 and V2), the required surface area is set to 2 m² per inhabitant. In case of combined treatment of greywater and faeces filtrate (SC2), the required surface area is increased to 2.5 m² per 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 of consultants.

Table 103: Material data for system components of a soil filter with upstream sedimentation tank

Component	Material			Source
Sedimentation tank	Concrete	29582	kg	Mall, 2005
(volume: 40 m³)	Steel, non-alloy	444	kg	
	Excavation	60	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	
	PE foil	1.4	kg/m²	Spiess-Wallbaum, 2002
	PES fleece	0.16	kg/m²	
Pipe DN50 (Feed)	PE	0.5	kg/m²	Length: Oldenburg, 2002 Weight: Simona, 2005
Pipe DN100 (Drainage)	PE	1.44	kg/m²	

* area for soil filter: 2 m²/pe in scenarios SV2 and V2, 2.5 m²/pe in scenario SC2

Urine tanks

Urine is interim stored in underground tanks made of glass fibre reinforced plastic. The required volume of the tanks is calculated from the daily amount of urine (1.5 L/(pe*d)), the separation efficiency (70%), and a minimum holding time of 14 days. The urine is withdrawn from the tanks by a suction vehicle and transported to the ozonation plant for micropollutant oxidation. The ozonation plant is excluded from the construction inventory due to lack of appropriate material data. From the plant, the urine is carried to the nearby farms for storage (> 6 months) and subsequent application on the fields. Storage tanks at the farms are not included in the inventory data, because farmers are assumed to use existing tanks for liquid manure to store the urine.

Table 104: Material data of interim storage tanks for separated urine

Component	Material	Weight [kg]	Qty
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 in the settlement. At present, a reliable and efficient system is not commercially available. Thus, the material demand for a separator has to be roughly estimated (Table 105). 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 to the greywater sewer and treated with greywater. The separated solid matter is stored in dewatering containers before it is brought to the composting plant by truck.

Vacuum system

The vacuum system for the collection and transport of faeces comprises of three vacuum stations which are located in the settlement. A vacuum station consists of 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 all system parts is estimated due to lack of appropriate material data (Table 106).

Table 105: Material data for a solid-liquid separator

Component	Material			Qty	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

Table 106: Material data for a vacuum station

Component	Material			Qty	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
- Central heat and power plant for biogas usage
- Service building

For pretreatment and hygienisation, two tanks with 25 m³ each are provided, which can simultaneously act as buffer tanks. For the main biogas unit, available data describes the material demand for a biogas plant with a digester volume (DV) of 600 m³ (Edelmann et al., 2001; Ronchetti et al., 2002). The post-digestion tank has a volume of 250 m³, and

two gas storage tanks with 225 m³ each can store the generated biogas. This data is adjusted to the amount of substrate load in this study via linear extrapolation.

In scenarios with urine separation (SC and SV), there is a daily load of ca. 30 m³ of digester substrate (faeces + biowaste). This load increases to 36 m³/d if urine is not separated (urine + faeces + biowaste, scenarios V). With a DV of 600 m³, calculated average residence times would be 20 or 17 days for the substrate inputs in this study, respectively. However, the targeted average residence time for the digestion process is 30 days. To account for the increased demand for DV at higher residence times, the material demand is increased proportionally for all materials related to the DV (pretreatment tank, digestor, post-digestion tank). Resulting factors are 1.5 in case of digestion of faeces and biowaste (30 m³/d for 30 days = 900 m³ DV = 1.5 * 600 m³) and 1.8 in case of digestion of urine, faeces, and biowaste (36 m³/d for 30 days = 1080 m³ DV = 1.8 * 600 m³).

The two CHP engines have a maximum power output of 25 kW_{el}/ 50 kW_{th}. For the housing of CHP engines and other equipment, a service building is provided.

Table 107: Material data for a biogas plant with 600 m³ digester volume

Component	Material			Qty	Source
Tanks (25 m³)	Concrete*	10000	kg	2	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 (600 m³) incl. gas storage (450 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		

**for these materials, linear factors are applied additionally to the listed values to account for increased demand of digester volume: factor 1.5 with urine separation (scenarios SV), factor 1.8 without urine separation (scenarios V)*

12.7 Transport distances

Table 108: Transport distances in LCA studies

in km	1)	2)	3)	4)	5)	This study
Fertilizers						
Mineral fertilizer		1200		600 + 100*		**300**
Sewage sludge		25	30			**20**
Compost to composting						**20**
Compost to farms		2	(30)			**20**
Faeces to composting						**20**
Digester residual	(20)	25	15 (30)		30	**20**
Urine to treatment						**5**
Urine to farms		8			30	**20**
Waste						
Sludge to incineration	30		20			**30**
Biowaste to incineration						**30**
Chemicals						
Flocculation	50	1200		600 + 100*		**300**
Construction materials						
Concrete to site			57	50		**50**
Concrete to disposal						**50**
Others to site			57	200 + 100**		**300**
Others to disposal						**100**

1) Frischknecht et al., 1996
2) Bengtsson et al., 1997
3) Reckerzügl, 1997
4) Frischknecht and Jungbluth, 2002
5) Jönsson et al., 2004
** for basic chemicals (train + truck)*
*** for plastic and metals (train + truck)*

12.8 Composition and production of mineral fertilizer

12.8.1 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 109 and Table 110 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 amounts of fertilizer applied in Germany in 1998/99 (Drescher-Hartung et al., 2001). Fertilizers with marginal market shares are neglected.

The multi-nutrient fertilizers (NP, NPK, and PK) 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). Potassium chloride (KCl) is considered as the most important K-fertilizer. 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 109: 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		18	
Singlesuperphosphate		20		2
Triplesuperphosphate		48.5		11
Raw phosphate		26		3
NP-fertilizer (20/20/0)	20	20	7	29
NPK- fertilizer (15/15/15)	15	15	9	40
PK- fertilizer (0/15/20)		15		15

Sources: Patyk and Reinhardt, 1997; Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001

Table 110: Nutrient content and shares of lime and K-fertilizer

Fertilizer	Nutrient content [%]		Market share [%]	
	K_2O	CaO	rel. to K	rel. to CaO
Potassium chloride	60		100	
$CaCO_3$ (lime)		54.3		85
Quick lime		97		15
Calciumammoniumnitrate (CAN)		21		--

Sources: Patyk and Reinhardt, 1997; Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001

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 (Boysen, 1992). Recent research results from the federal agency for agricultural research indicate considerable concentrations of uranium in certain fertilizers (Kratz, 2004; Fink, 2005). Uranium is present in raw phosphate rock, and is further enriched during the production process of superphosphate or triplesuperphosphate, resulting in concentrations of 13 – 191 mg U/kg in these fertilizers. Due to the limited output of uranium via groundwater or agricultural products, an accumulation of toxic uranium in agricultural soils is likely, which may result in an enhanced transfer into agricultural products.

Table 111: 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	3.3	0.25	8.7	4	3.8	0.01	21.4	< 1	38.3
Urea	0.04	0.13	0.5	0.5	0.7	0.01	0.6	< 1	1.9
Ureaammoniumnitrate		0.03	1.3	6.3	0.3		0.2	< 1	2.3
Singlesuperphosphate	3.7	10.8	114	17.2	28.8	0.02	18.5	138	236
Triplesuperphosphate	3.7	26.8	288	27.3	36.3	0.04	12	138	489
Raw phosphate	3.6	7.8	168	15.6	15.6	0.07	1.3	44	199
NP-fertilizer (20/20/0)		9.15	91.4	21.5	18	0.02[2)]	5.5	93	151
NPK-fertilizer (15/15/15)		3.78	45.8	11.3	10.9	0.06[2)]	14.8	14	116
PK-fertilizer (0/15/20)		7.98	191	19.3	19.9	0.08[2)]	14.4	93	152
Potassium chloride	0.01	0.08	3.5	2.9	1.5	0.02	0.5	< 1	3.7
Lime ($CaCO_3$)	0.05	0.3	7.5	8.2	6.1	0.04	5.9	< 1	41.2
Quick lime		0.1	19.2	11.1	6		2.8	< 1	15.8

1) Vogt et al. (2002)
2) Drescher-Hartung et al. (2001)
3) Estimations according to Kratz (2004)

Table 111 lists the average heavy metal content of mineral fertilizers. The literature data is essentially based on the work of Boysen (Boysen, 1992). The values for uranium are mean values derived from concentration ranges (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 112), the data from Table 111 is connected with the nutrient content and market shares of the respective fertilizers. Data for arsenic in multi-nutrient fertilizers is estimated via single-nutrient fertilizers.

Table 112: Calculated 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

Sources: Patyk and Reinhardt, 1997; Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001

Recently, updated values for heavy metal contaminations of several mineral fertilizers sold in the EU were published (UBA, 2007a). These values from the database of the FAL (Federal Agricultural Research Centre) indicate substantial changes in relation to the Boysen study. If the updated values are taken as the basis for calculation (Table 114), the contamination with some heavy metals decreases significantly (Cr, Ni, Pb), while the content of Cu (added as micronutrient) and Zn increases. However, the UBA study suffers from a lack of adequate data (small number of samples), and the published values are not considered as representative for the national average by the authors. Hence, the present study is based on the comprehensive results of the Boysen study despite its relatively high age of 15 years.

Table 113: Updated concentrations of heavy metals and As in mineral fertilizers

Fertilizer	Concentrations of heavy metals and As [mg/kg dry matter]								
	As	Cd	Cr	Cu	Ni	Hg	Pb	U	Zn
Calciumammoniumnitrate	0.31	0.17	2.9	4.0	2.3	0.01	16.0	0.23	32.3
Urea		0.03	0.33	0.4	0.26	0.07	0.09		1.7
Singlesuperphosphate	3.9	3.3	23.9	24.8	16.3	0.04	20.9	72.2	93.6
Triplesuperphosphate	10.7	14.4	131	5.2	17.6	0.09	10.1	197	159
NP-fertilizer (20/20/0)	11.2	7.8	63	4.6	16.4	0.12	5.1	65.2	121
NPK-fertilizer (15/15/15)	2.0	4.5	18.2	172	5.5	0.02	6.1	22.6	283
PK-fertilizer (0/15/20)	5.2	4.5	79.5	12.3	17.5	0.07	4.2	87.6	128
Potassium chloride	0.25	1.0	6.5	1.6	2.3	0.09	3.1	0.56	2.2

Source: UBA, 2007a, based on small number of samples from FAL database

Table 114: Mean concentrations of heavy metals and As for average mineral fertilizers, calculated with values based on UBA study (2007)

Values in mg/kg nutrient	As	Cd	Cr	Cu	Ni	Hg	Pb	U	Zn
N-fertilizer (as N)	5.7	5.8	39.3	120.4	13.9	0.1	38.2	36.8	284.6
P-fertilizer (as P_2O_5)	30.0	32.3	269.8	483.8	63.6	0.3	32.7	301.8	1.126.9
K-fertilizer (as K_2O)	0.1	1.7	10.8	2.7	3.8	0.2	5.2	0.9	3.7
Lime (as CaO)	0.1	0.5	14.7	14.6	10.5	0.1	9.7	1.0	66.9

Sources: Patyk and Reinhardt, 1997; Hackenberg and Wegener, 1999; Drescher-Hartung et al., 2001; UBA, 2007a

12.8.2 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 115 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 115a: Life cycle inventories of mineral fertilizer production

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	0.022	0	0
CO2, fossil	*kg*	2820[1)]	1117	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

Table 115b: Life cycle inventories of mineral fertilizer production (ctd)

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	4366[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	1310[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, i.e. including CO_2 emissions from urea hydrolysis in soil: 2980 kg if urea has 10% market share (rel. to applied N)
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)
Sources: Patyk and Reinhardt, 1997; Gaillard et al., 1997 for aquatic emissions

Different production processes and countries of origin are considered in relation to their market share in Germany 1993 (aquatic emissions: 1998/99). Transport of the fertilizers to the farms and emissions during application are not included, and neither is the construction and maintenance of infrastructure.

Regarding the data quality, recent changes in market shares or production processes are not accounted for in this data set. However, the data for the emissions of fluoride and phosphate during P-fertilizer production presented in Gaillard et al., 1997 is based on production data from 1992 (Audsley et al., 1997). The emission factors are very high compared to more recent data (Table 116) and have a strong influence on the impact

assessment due to the high toxicity of fluoride. Hence, it is decided to adjust these emission factors in view of the advances in exhaust gas and wastewater treatment.

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. Removal efficiencies of 99% are achievable with regard to HF emissions (Wiesenberger, 2002). Literature data shows a wide range of emission factors for fluoride and phosphate emissions (Table 116) depending on production route, emission control measures, and environmental legislation. Hence, moderate emission factors for the present inventory are estimated based on the collected data, assuming state-of-the art exhaust air cleaning and wastewater treatment according to European standards.

Table 116: Emissions of fluoride and phosphate in P fertilizer production

[mg/kg P]	**F (as fluorides)**		**PO_4**	**Remarks**
Source	**Air**	**Water**	**Water**	
Audsley et al., 1997	460	167000	138000	Data from 1992 for triplesuperphosphate
HELCOM, 1996		690	351	Recommendations for critical load
Wiesenberger, 2002	13.33[1]/217[1]	2680	4141	Plant data, air emissions for triplesuperphosphate/NPK on mixed acid route
Wiesenberger, 2002	67[1]	290	530	Plant data for NPK fertilizer with nitro-phosphate route
EPA, 1995	874			Emission factors for superphosphate production (US)
EPA, 2000	11 – 227			EPA standards for new plants (depending on production route)
EC, 2007	6 – 309	4580 – 71000	4912 – 9123	Reported emissions in phosphoric acid production
IFC, 2007	23	1603	3677	Industry benchmark for nitro-phosphate route
IFC, 2007	167	250		Industry benchmark for mixed acid route
This study	**50[1]**	**3000**	**10000**	**estimated**

1) as HF

12.9 Data from Life Cycle Inventory

This section presents selected data from the Life Cycle Inventory. The presented data constitutes the basis for the results of LCI as presented in chapter 5.1. In particular, the following information is listed:

1. Input-output balance of electrical energy for operation (Table 117)

2. Calculated plant-available nutrient content for mineral and organic fertilizers (Table 118)
3. Calculated effluent volume, loads, and concentrations from wastewater and greywater treatment (Table 119)
4. Calculated heavy metal emissions to surface waters and agricultural soil (Table 120)

Table 117: Input-output balance of electrical energy for operation

in kWh/(pe*a)	R	R_{min}	R_{agri}	V1	V2	V3	SV1	SV2	SV3	SC1	SC2	SC3
INPUT												
Drinking water	19	21.2	19	15.5	15.5	14.6	15.5	15.5	14.6	19	19	14.6
Pumping/Vacuum	0	0	0	16.0	15.8	17.0	16.0	15.8	17.0	3	1.9	5.6
Composting	2.4	2.4	2.4	0	0	0	0	0	0	5.2	6.4	5.4
Digestion	0	0	0	14.1	14.8	14.1	15.0	15.8	15.0	0	0	0
Wastewater treatment	24	28.3	24	14.7	1.6	19.4	14.9	1.6	19.6	19.1	2.1	25
Urine separation	0	0	0	0	0	0	5.8	5.8	5.8	5.8	5.8	5.8
SUM INPUT	**45.4**	**51.9**	**45.4**	**60.3**	**47.7**	**65.1**	**67.2**	**54.5**	**72**	**52.1**	**35.2**	**56.4**
OUTPUT												
Sewage gas/biogas	12.1	0	12.1	44.4	47.6	44.5	40.9	44	40.9	0	0	0
Incineration of biowaste	3.1	3.1	3.1	3.1	3.1	3.1	3.1	3.1	3.1	3.1	3.1	3.1
Incineration of sludge	0.7	1.0	0	0.2	0.4	0	0.2	0.3	0	0	0.4	0
*Feedstock**	0.2	0.2	0.2	0.7	1.0	1.1	1.1	1.4	1.5	1.0	1.3	1.4
SUM OUTPUT	**16.1**	**4.3**	**15.4**	**48.4**	**52.1**	**48.7**	**45.3**	**48.8**	**45.5**	**4.1**	**4.8**	**4.5**
NET ENERGY DEMAND	**29.3**	**47.6**	**30**	**11.9**	**-4.4**	**16.4**	**21.9**	**5.7**	**26.5**	**48**	**30.3**	**51.9**
SYSTEM EXPANSION	36	47.8	36.7	3.7	0	3.4	6.8	3.3	6.6	48	47.3	47.6

* energy credit from infrastructure (incineration of plastic materials)

Table 118: Calculated plant-available nutrient content of mineral and organic fertilizers

in g/(pe*a)	R	R_{min}	R_{agri}	V1	V2	V3	SV1	SV2	SV3	SC1	SC2	SC3
NITROGEN												
Compost	29	29	29	0	0	0	0	0	0	64	65	63
Urine	0	0	0	0	0	0	2254	2254	2254	2254	2254	2254
Digester sludge	0	0	402*	3241	3255	3247	325	323	326	0	0	0
Mineral fertilizer	3226	3226	2824	14	0	8	675	677	674	937	935	938
PHOSPHORUS												
Compost	119	119	119	0	0	0	0	0	0	250	254	246
Urine	0	0	0	0	0	0	256	256	256	256	256	256
Digester sludge	0	0	490*	718	724	719	393	398	394	0	0	0
Mineral fertilizer	605	605	115	6	0	5	75	70	74	218	214	222
POTASSIUM												
Compost	480	480	480	0	0	0	0	0	0	689	709	692
Urine	0	0	0	0	0	0	664	664	664	664	664	664
Digester sludge	0	0	10*	1644	1664	1678	490	500	507	0	0	0
Mineral fertilizer	1198	1198	1188	34	14	0	524	514	507	325	305	322

* sewage sludge (digested)

Table 119: Calculated effluent volume, loads, and concentrations from wastewater and greywater treatment

		R	R_{min}	R_{agri}	V1	V2	V3	SV1	SV2	SV3	SC1	SC2	SC3
Effluent volume	m³/(pe*a)	38.5	42.9	38.5	28.7	25.5	27.2	30.8	27.5	29.2	37.3	33.4	29
Effluent loads													
COD	kg/(pe*a)	2.0	2.0	2.0	0.9	1.3	0.9	1.0	1.4	0.9	1.2	1.7	0.9
NH_4-N	kg/(pe*a)	0.04	0.2	0.04	0.01	0.01	0.01	0.03	0.03	0.03	0.04	0.05	0.03
N total, inorg	kg/(pe*a)	0.3	1.7	0.3	0.08	0.16	0.1	0.15	0.23	0.17	0.3	0.6	0.3
N total	kg/(pe*a)	0.4	2.1	0.4	0.13	0.25	0.16	0.24	0.37	0.27	0.45	0.9	0.5
P	g/(pe*a)	30.4	364	30.4	16.4	82.2	7.6	20	85.8	11.2	27	135	10.8
Effluent concentrations													
COD	mg/L	52.3	47.0	52.3	32.7	52.5	32.3	32.0	50.5	31.5	31.7	50.7	31.3
NH_4-N	mg/L	1.1	4.9	1.1	0.3	0.4	0.3	0.9	1.1	0.9	1.2	1.6	1.2
N total, inorg	mg/L	8.7	38.9	8.7	2.8	6.1	3.6	5.0	8.4	5.9	7.6	16.7	10.9
N total	mg/L	10.8	48.6	10.8	4.4	9.9	5.8	7.8	13.4	9.3	11.9	26.7	17.5
P	mg/L	0.8	8.5	0.8	0.7	4.0	0.4	0.6	3.1	0.4	0.7	4.0	0.4

Note:
- in scenarios SV1+2+3, values include effluent from separate treatment of sludge liquor from digestion
- in scenarios SC1+2+3, faeces filtrate is co-treated with greywater

Table 120: Calculated heavy metal emissions to surface waters and agricultural soil

in mg/(pe*a)	R	R_{min}	R_{agri}	V1	V2	V3	SV1	SV2	SV3	SC1	SC2	SC3
Emissions to surface waters												
Cd	32	32	27	22	22	21	25	25	23	26	25	20
Cr	281	285	253	221	220	207	239	239	225	247	246	195
Cu	1429	1535	1403	1096	1460	1024	1137	1503	1060	1320	1755	886
Ni	379	388	357	293	293	274	335	335	317	322	322	261
Hg	9	9	4	2	2	2	3	3	3	4	4	4
Pb	295	306	272	247	237	234	270	260	257	280	267	225
*U**	8	10	8	7	8	5	9	10	6	10	7	11
Zn	5999	6405	5966	4203	3361	3928	4773	3942	4471	5036	4031	3450
Emissions to agricultural soil												
Cd	88	88	100	21	21	21	25	25	26	45	45	46
Cr	1280	1280	1555	279	281	282	370	373	373	613	619	619
Cu	941	941	8706	1590	1661	1357	1389	1450	1192	1354	1440	1360
Ni	340	340	748	240	246	249	179	183	185	295	302	293
Hg	8	8	20	15	15	15	12	12	12	13	13	15
Pb	567	567	1388	305	314	309	295	303	299	383	393	385
*U**	664	664	249	6	5	0	98	98	94	227	224	230
Zn	6475	6475	23278	9059	9212	8628	7095	7212	6773	8961	9153	8965

* uranium in water from energy production (nuclear power), uranium in soil from mineral phosphate fertilizer

12.10 Valuation results

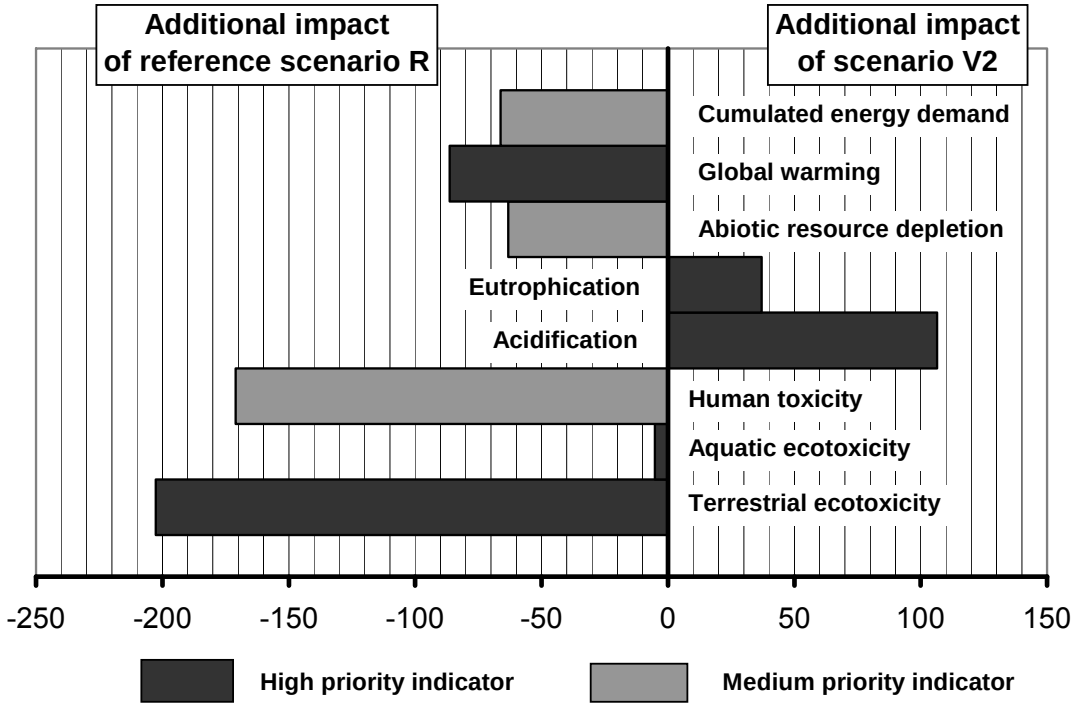

Figure 97: Comparison of scenario R and V2 with valuated indicator results in T diagram

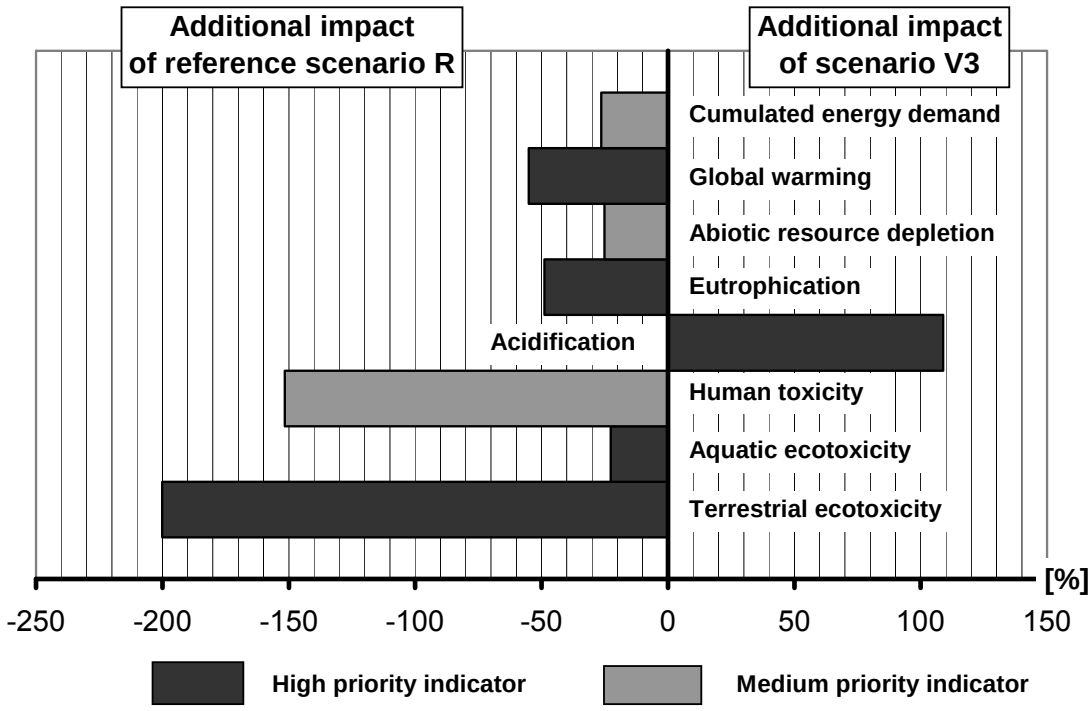

Figure 98: Comparison of scenario R and V3 with valuated indicator results in T diagram

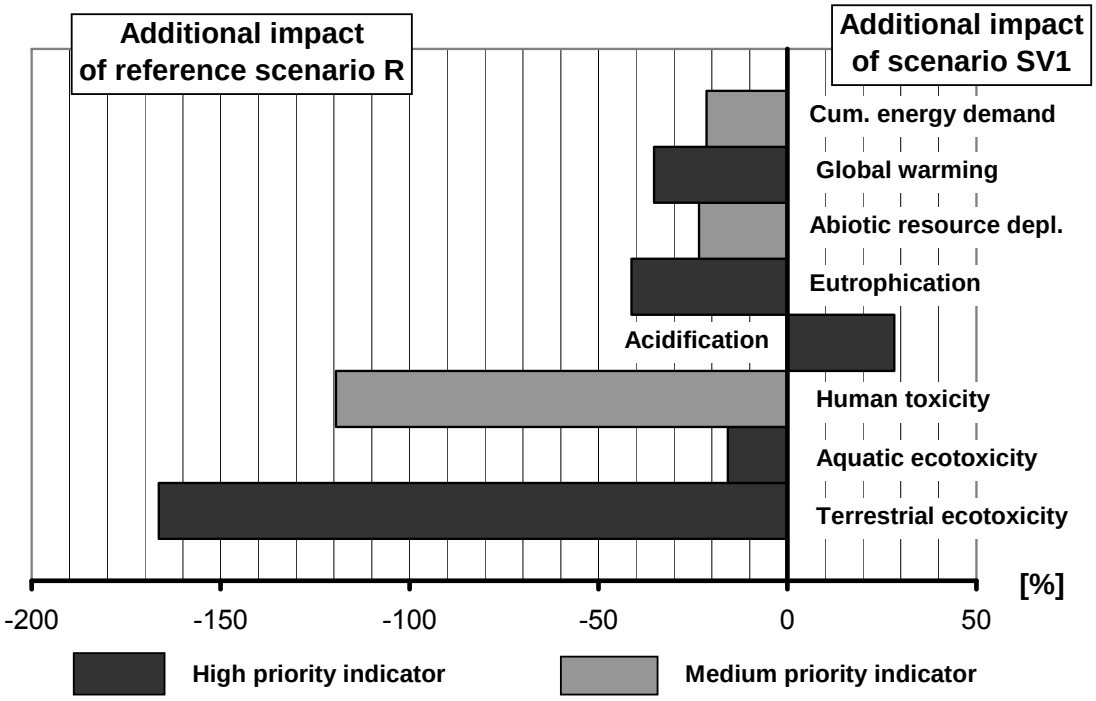

Figure 99: Comparison of scenario R and SV1 with valuated indicator results in T diagram

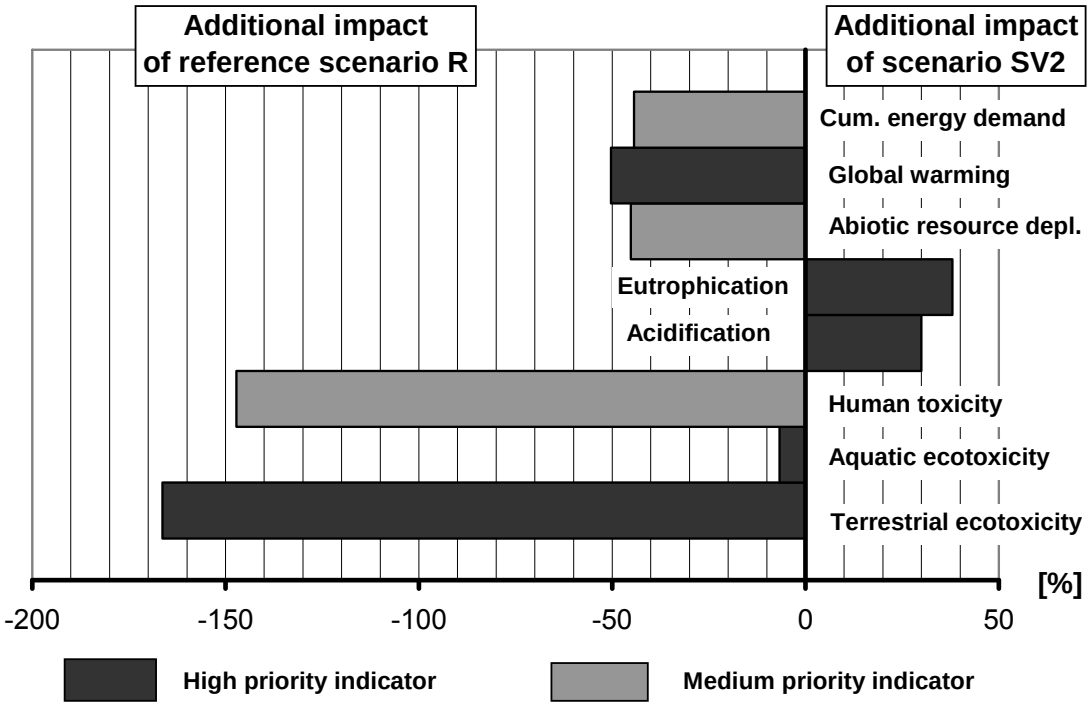

Figure 100: Comparison of scenario R and SV2 with valuated indicator results in T diagram

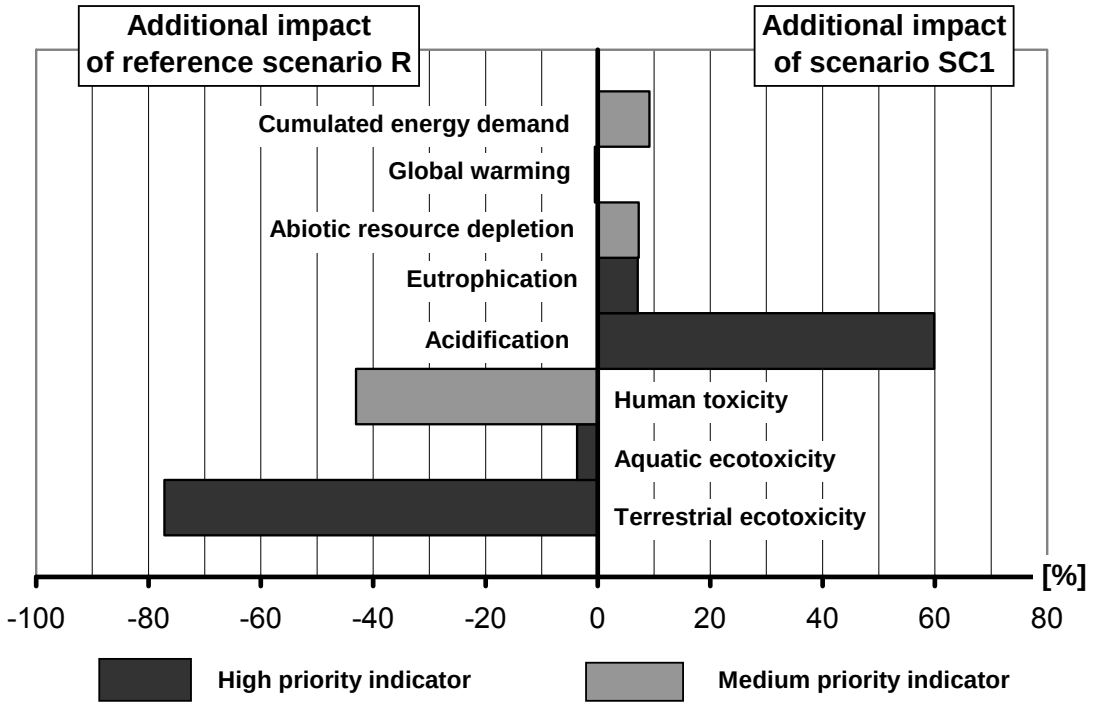

Figure 101: Comparison of scenario R and SC1 with valuated indicator results in T diagram

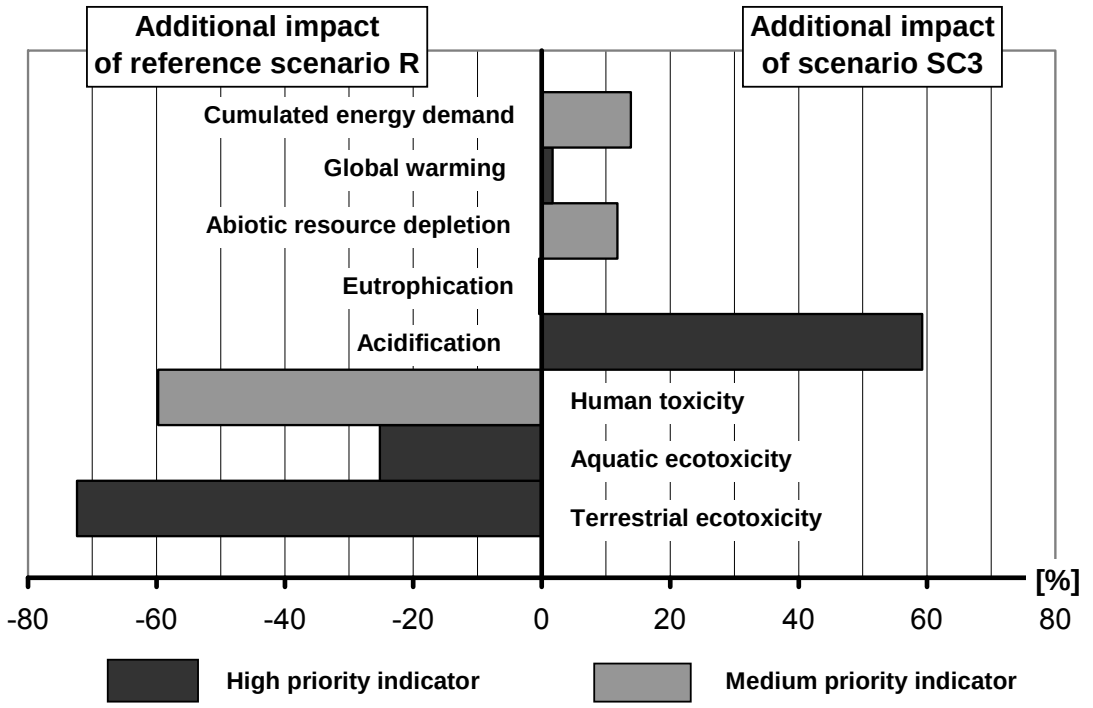

Figure 102: Comparison of scenario R and SC3 with valuated indicator results in T diagram

12.11 List of Tables in Annex

12.12 List of Figures in Annex

12.13 Literature in Annex

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