RESOURCE EFFICIENCY OF URBAN SANITATION SYSTEMS: A COMPARATIVE ASSESSMENT USING MATERIAL AND ENERGY FLOW ANALYSIS

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Things alter for the worse spontaneously,
if they be not altered for the better designedly.

Francis Bacon (1561-1626)

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Abstract

Within the framework of sustainable development it is important to find ways of reducing natural resource consumption and to change towards closed-loop management. As in many other spheres increased resource efficiency has also become an important issue in sanitation. Particularly nutrient recovery for agriculture, increased energy-efficiency and saving of natural water resources, can make a contribution to more resource efficient sanitation systems. To assess the resource efficiency of alternative developments a systems perspective is required.

The present study applies a combined cost, energy and material flow analysis (ceMFA) as a system analysis method to assess the resource efficiency of urban sanitation systems. This includes the discussion of relevant criteria and assessment methods. The main focus of this thesis is the comparative assessment of different systems, based on two case studies; Hamburg in Germany and Arba Minch in Ethiopia. A range of possible system developments including source separation (e.g. diversion of urine or blackwater) is defined and compared with the current situation as a reference system. The assessment is carried out using computer simulations based on model equations. The model equations not only integrate mass and nutrient flows, but also the energy and cost balances of the different systems. In order to assess the impact of different assumptions and calculation parameters, sensitivity analyses and parameter variations complete the calculations. Based on the simulations, following general conclusions can be drawn:

- None of the systems show an overall benefit with regard to all investigated criteria, namely nutrients, energy, water and costs. Yet, the results of the system analysis can be used as basis for decision making if a case-related weighting is introduced.
- The systems show varying potential for the recovery of nutrients from (source separated) wastewater flows. For the case study of Hamburg up to 29% of the mineral fertiliser could be substituted by nutrients recovered from wastewater; for the case study of Arba Minch this substitution amounts to a maximum of 16%.
- Factors such as the transport of source separated flows or complex nutrient recovery processes can result in an increasing energy demand. However, source separation and recovery processes can also lead to energy reduction, for example, by urine diversion (minus 12% for the case of Hamburg) or by the use of biogas from anaerobic treatment plants (minus 38% for the case of Arba Minch). The energy efficiency depends on determinant parameters, e.g. the amount of co-digested organic waste. The impact of these parameters can be simulated in the model.

- Source-separating wastewater systems can reduce the use of natural water resources, for example, by reduced flush water consumption or greywater recycling.
- The integration of cost estimates with material and energy flow analyses, allows a cost-effectiveness appraisal of the system developments. Assumptions such as whether the costs refer to a new development or the modification of existing infrastructure have a major impact on the cost comparison. Where the sanitation system is improved, there is invariably an increase in costs when compared to the current situation. But in addition, financial benefits can be generated.

For each case study, a discussion of the driving forces, preconditions and starting points for implementation, complements the comparative assessment. In addition, potential obstacles for transformation are discussed. The study shows that the method of using combined cost, energy and material flow analysis yields purposeful insights into the resource efficiency of alternative sanitation systems. This can contribute comprehensively to system analysis and decision support.

Kurzfassung

Im Kontext einer nachhaltigen Entwicklung wird es immer wichtiger Möglichkeiten zu finden, die vorhandenen natürlichen Ressourcen zu schonen und Potentiale zur Kreislaufwirtschaft aufzuzeigen und umzusetzen. Auch in der Abwasserwirtschaft ist die Steigerung der Ressourceneffizienz zu einem aktuellen Thema geworden. Insbesondere die Rückgewinnung von Nährstoffen, die Steigerung der Energieeffizienz und die Schonung der natürlichen Wasserressourcen können einen Beitrag zu ressourceneffizienteren (Ab)Wassersystemen leisten. Hierfür ist es notwendig, eine systembezogene Betrachtungsweise einzunehmen, um mögliche alternative Systeme umfassend bewerten zu können.

In der vorliegenden Arbeit wird unter Anwendung von kombinierten Kosten-, Energie- und Stoffstromanalysen (ceMFA) die Ressourceneffizienz verschiedener städtischer Abwassersysteme systemanalytisch untersucht. Neben einer Diskussion relevanter Kriterien und Bewertungsmethoden bildet die vergleichende Analyse verschiedener Systeme den Schwerpunkt dieser Studie. Für zwei Fallbeispiele - Hamburg in Deutschland und Arba Minch in Äthiopien - werden eine Reihe von möglichen Systementwicklungen inklusive Teilstrombehandlungen (z.B. Separation von Urin oder Schwarzwasser) definiert. Als Referenzsystem dient jeweils das gegenwärtig vorhandene Wasser- und Abwassersystem. Für die Analyse werden in einem Simulationsprogramm Modellgleichungen aufgestellt, die Massen- und Nährstoffströme, Energiebilanzen sowie Kostenfunktionen der unterschiedlichen Systeme integrieren. Die Berechnungen werden abgerundet durch Sensitivitätsanalysen und Parametervariationen, die den Einfluss verschiedener Annahmen und Berechnungsparameter untersuchen. Aus den Simulationen lassen sich folgende generelle Schlussfolgerungen ziehen:

- Keines der Systeme zeigt einen Gesamtnutzen in Bezug auf alle untersuchten Kriterien, d.h. Nährstoffe, Energie, Wasser und Kosten. Die Ergebnisse der Systemanalyse können jedoch mit Hilfe einer fallbezogenen Gewichtung als Basis für eine Entscheidungsunterstützung dienen.
- Die untersuchten Systeme besitzen unterschiedliches Potential zur Nährstoffrückgewinnung aus Abwasser(teil)strömen. Für das Fallbeispiel Hamburg könnten bis zu 29% des Mineraldüngereinsatzes reduziert werden, für Arba Minch beträgt die maximale Reduzierung 16%.
- Faktoren wie der Transport von Teilströmen oder aufwändige Rückgewinnungsverfahren können teilweise zu einer Erhöhung des Energiebedarfs führen. Auf der anderen Seite können jedoch auch energetische Einsparungen durch die Einführung von Teilstrombehandlung erzielt werden, wie z.B. durch

Urinseparation (minus 12% im Fall Hamburg) oder die Nutzung von Biogas aus anaeroben Behandlungsanlagen (bis zu 38% im Fall Arba Minch). Die Energieeffizienz hängt von determinierenden Parametern ab, wie z.B. der Menge an mitvergärten organischen Abfällen, deren Einfluss im Modell simuliert werden kann.

- Die Nutzung natürlicher Wasserressourcen kann durch teilstromorientierte Abwassersysteme reduziert werden, z.B. durch reduzierte Spülwassermengen oder Grauwasserrecycling.
- Die Integration der Kostenschätzungen in die Stoffstrom- und Energieanalysen ermöglicht eine Abschätzung der Kostenwirksamkeit der Systemanpassungen. Dabei zeigt sich, dass die Randbedingungen wie Neubau oder Implementierung im Bestand einen erheblichen Einfluss auf den Kostenvergleich besitzen. Zum Teil erhöhen sich unter den im Modell gewählten Annahmen die Kosten der Systeme im Vergleich zur gegenwärtigen Situation. Eine differenzierte Analyse zeigt jedoch die Abhängigkeit der Kosten von bestimmten Parametern sowie die Möglichkeit der Generierung eines finanziellen Nutzens.

Die vergleichende Bewertung wird komplementiert durch eine Diskussion verschiedener Treiber, Voraussetzungen und Möglichkeiten zur Implementierung ressourceneffizienter Systeme bezogen auf die jeweilige Fallstudie. Auch mögliche Hindernisse bei der Umsetzung werden aufgezeigt und diskutiert. Die Studie zeigt, dass die Methodik der kombinierten Stoffstrom- und Energieanalysen gezielte Einblicke in die Ressourceneffizienz verschiedener Abwassersysteme geben kann und einen umfassenden Beitrag zur Systemanalyse und Entscheidungsfindung leisten kann.

Table of Contents

Tables

Figures

Glossary

arbour loo	shallow pit used for defecation, ash and soil is added and a tree is planted on the pit after it has filled, while a new pit is dug and the latrine relocated (see also Morgan, 2007)
biowaste	used in this study as synonym for organic waste
blackwater	toilet wastewater
greenfield	area, where no previous facilities exist
fossa alterna	alternating, waterless double pit designed to make compost (see also Morgan, 2007)
greywater	wastewater from households without wastewater from toilets, usually including kitchen wastewater and water from washing, bathing, cleaning
Hamburg Wasser	umbrella organisation of Hamburg Waterworks ltd. and Hamburg Public Sewage Company; responsible for the public water and wastewater infrastructure and operation of services in the Metropolitan Region of Hamburg
hinterland	rural areas surrounding and delivering products to urban areas
material	in MFA terminology material stands for both, substances (e.g. chemical elements or compounds) and goods (e.g. drinking water or wastewater)
nightsoil	human excreta collected in containers or buckets
parameter	data provided as input to the MFA to calculate the value of a variable
process	processes are key components of an MFA and specify any transport, transformation, storage and change of value of materials and goods (Baccini and Brunner, 1991)
ROSA project	project on "Resource-Oriented Sanitation Concepts for Peri-Urban Areas in Africa", funded by the European Union within the scope of the EU 6th framework programme as a Specific Target Research Project (STREP), Sub-Priority "Global Change and

	Ecosystems", contract no. 037025-GOCE, duration: 1 October 2006 – 31 March 2010. http://rosa.boku.ac.at/
sanitation	in this study, the term sanitation refers to the management of human excreta and other liquid household wastes. The terms sanitation and wastewater management are used interchangeably
variable	used in MFA terminology to define flows (input, output and inner flows), stock rates and stock rates changes
yellowwater	urine and flushwater

Abbreviations and symbols

AHP	Analytical Hierarchy Process
AMU	Arba Minch University
build.	building
cap.	capita (also used as population equivalent)
ceMFA	cost, energy and material flow analysis
CH_4	methane
CHP	combined heat and power unit
CO_2	carbon dioxide
COD	chemical oxygen demand
CSO	combined sewer overflow
d	day
DL	daily load
EIA	Environmental Impact Assessment
ETB	Ethiopian Birr (exchange rate: OANDA, 2009) *(1 December 2009: 1 ETB = 0.08 US$ = 0.05 €)*
€	Euro (exchange rate: OANDA, 2009) *(1 December 2009: 1 € = 1.50 US$ = 18.77 ETB)*
FAO	Food and Agriculture Organisation
g	gram
GDP	gross domestic product
GIS	geographic information system
GNP	gross national product
GWh	gigawatt hours
η	efficiency ratio
h	hours
hh	household
inh	number of inhabitants
J	Joule
K	potassium

kg	kilogram
km	kilometre
kWh_{el}	kilowatt-hours (electricity)
kWh_{PE}	kilowatt-hours (primary energy)
kWh_{th}	kilowatt-hours (thermal energy)
LCA	life cycle assessment
LHV	lower heating value (net calorific value)
LU	livestock unit
MAP	magnesium ammonium phosphate ($MgNH_4PO_4$)
MBR	membrane bioreactor
MDG	Millennium Development Goals
MFA	material flow analysis
mg	milligram
MMFA	mathematical material flow analysis
MJ	megajoule (3.6 MJ = 1 kWh)
MSE	micro and small enterprise
N	nitrogen
NH_4-N	ammonium nitrogen
N_2O	nitrous oxide
NPV	net present value
P	phosphorus
p	person
S	sulphur
SI	international system of units
stdev	standard deviation
t	tonne
tc	transfer coefficient
TOC	total organic carbon
toe	tonne of oil equivalent (1 toe = 11.63 MWh)
TS	total solids (suspended and dissolved)

TWh	terawatt-hour, i.e. 10^{12} watt-hours
UDDT	urine-diverting, dry toilet
US$	U.S. Dollar (exchange rate: OANDA, 2009) *(1 December 2009: 1 US$ = 0.67 € = 12.49 ETB)*
VIP	ventilated improved pit latrine
VS	volatile solids
WASH	water, sanitation and hygiene
WWTP	wastewater treatment plant
y	year

1 Introduction

Sanitation is a basic need and is considered to be essential for hygiene, disease prevention and environmental protection. One of the primary focuses of sanitation is on the appropriate management of human excreta, i.e. toilet wastes, and other liquid household wastes. But sanitation should go beyond dealing with wastes and should acknowledge the value of waste as a resource. Therefore, tomorrow's sanitation systems need to be able to manage the available resources efficiently.

This study aims to contribute to the achievement of increased resource efficiency in sanitation. The following sections provide some background information, as well as the specific objectives of this study. Chapter 1 closes with an overview of the structure of this thesis.

1.1 Background

The world is clearly urbanising with about 70% of the world's population expected to live in urban areas by 2050 (UN, 2008). Along with increasing urbanisation, there is a need to adapt the urban infrastructure to provide services that contribute to sustainable development. Sanitation and wastewater infrastructure has not been given high priority in many regions of the world. Current statistics (e.g. WHO and UNICEF, 2008) suggest that more than 20% of the world's urban population do not have access to improved sanitation facilities, not to mention the lack of appropriate operation, treatment and disposal. But even in those parts of the world where full sanitation coverage is achieved and where facilities for wastewater treatment exist, critics argue that the efficiency and viability of many of the current systems is questionable; limitations such as restricted flexibility, waste of drinking water, high energy demand for removal of organic matter and nutrients, as well as high operation and rehabilitation costs need to be overcome (Otterpohl et al., 1997; Riße and Herbst, 2004; Wilsenach et al., 2003).

Based on historic decisions to mix and discharge, conventional sanitation systems break the link in anthropogenic nutrient cycles and follow a so-called "waste approach" (Czemiel Berndtsson, 2004). In the 19th century, nutrients in excreta turned from resources for agriculture (Salviati et al., 1865) into pollutants (Erismann, 1882). Since then, wastewater treatment has characterised by successive adaptations of technologies to keep up with stricter treatment requirements (i.e. following a “process-thinking”)[1]. Efficient resource management has not been a primary objective in sanitation in the past century, but this line of thinking is slowly emerging from discussions about sustainable sanitation systems (Dockhorn, 2007; Larsen et al., 2007). Therefore, the time has come to challenge the current systems, to move away from process-thinking and to shift towards system-thinking. This can eventually lead to the re-establishment of the link between sanitation and agriculture.

Innovative concepts for improved resource management in sanitation, which suit different socio-economic and geographic framework conditions, exist (Otterpohl et al., 1999). Source separation of different flows, instead of mixing wastewater, often plays a fundamental role in these concepts. Pilot implementations have been installed world-wide (DWA (ed.), 2008; SuSanA (ed.), 2010). These include a variety of different technologies and approaches. Notwithstanding the diversity of these approaches, their common aim is to fulfil sanitation needs, not only focusing on hygiene, health and environmental protection, but to contribute to a more efficient management of our resources.

It is often presumed that source control and nutrient recovery from wastewater can contribute to more efficient resource management. The starting point of this thesis is the need for transparent and holistic systems analysis methods, to test this hypothesis and to eventually support decision making.

1.2 Objectives of the study

The focus of this study is on the concept of resource efficiency in sanitation. Other primary objectives of sustainable sanitation systems, such as environmental protection, are only touched upon. Yet, decision making situations must also always take economic consequences into account. It is therefore important to identify the relationships and trade-offs that exist between the environmental and the economic performance of different systems. This study includes these considerations by integrating economic aspects into the concept of resource efficiency.

[1] While the first wastewater treatment technologies primarily aimed at the removal of suspended matter, standards evolved that first required the removal of organic matter and later included nutrient removal. Current developments suggest that in the near future micropollutants will be added to the list of pollutants that need to be reduced.

The main aim of this thesis is to contribute to systems analysis and strategic planning in sanitation. Two primary objectives are:

1. The development of a method to assess the resource efficiency in urban sanitation.

2. The application of this method to two case studies in order to assess different sanitation systems aimed at nutrient recovery.

The outcomes of this study are expected to contribute to increased knowledge of the shortcomings of conventional sanitation systems, as well as a better understanding of possible alternative solutions. Eventually, this can enhance the further development and improvement of alternative systems.

Two case studies, namely Hamburg in Germany and Arba Minch in Ethiopia, are selected for a detailed analysis of different possible systems. The case studies represent socio-economic and geographic backgrounds that are very distinct from each other. Also the level of infrastructure development is very different in those two towns. Therefore, this study embraces a wide range of sanitation options in different contexts and tries to grasp resource efficiency from a global, but also a local perspective. The analysis aims at bridging the differences in concepts and technologies by using the same assessment method. What is common in the two case studies, though, is that both towns are currently involved in their first implementations of alternative sanitation options, including source-separating facilities. Analysing the potentials for improved resource efficiency is therefore very applicable to both towns.

Several research questions build the framework for achieving the objectives of this study:

- What are relevant criteria when considering resource efficiency with regard to sanitation, particularly in the context of Germany and Ethiopia?
- Which methods are available to assess resource efficiency in sanitation?
- How can a suitable method be best adapted or developed for the two case studies, Hamburg and Arba Minch?
- Which systems are expected to increase the resource efficiency of urban sanitation in the respective settings?
- How do the selected systems score with regard to the different criteria of resource efficiency?
- Which are critical factors impacting on resource efficiency and how can the systems be adapted accordingly?
- Which additional issues need to be considered for a transformation towards more resource efficient systems?

1.3 Outline of the report

The next chapter (Chapter 2) deals with resource efficiency from a theoretical perspective, to provide a framework for the further analysis. With regard to sanitation, resource efficiency can be related to the criteria water, nutrients, energy and costs. Chapter 2 highlights the relevance and significance of these criteria in the context of Germany and Ethiopia.

Chapter 3 outlines the method of this study. After a brief overview of available assessment methods, the combined cost, energy and material flow analysis (ceMFA), which is used in this study, is illustrated. The chapter also includes a presentation of the two case studies, Hamburg and Arba Minch and introduces the selected sanitation systems.

The results of the resource efficiency assessment of the Hamburg systems and the Arba Minch systems are presented in Chapters 4 and 5 respectively. The relative performance of the different systems is analysed complemented by uncertainty and sensitivity analyses. Furthermore, parameter variations and modifications of the systems are studied. The assessment is completed by a discussion of possible system transformation processes.

In Chapter 6 the method as well as main results of the assessment are critically reviewed. The chapter finishes with a discussion about the integration of resource efficiency assessments into planning and decision making.

Finally, the main outcomes of this work are rounded up in a brief conclusion (Chapter 7).

2 Resource efficiency in sanitation

This chapter provides the theoretical framework for the study. First, the concepts of sustainability and resource efficiency are discussed in connection to water and sanitation. Relevant criteria used in the assessment of the case studies are introduced. Furthermore, the selected criteria water, nutrients, energy and costs are reviewed with a special focus on Germany and Ethiopia to provide a basis for the assessment.

2.1 The concept of sustainability in the context of sanitation

Sanitation implies the need to not only look at particular technologies but to take a systems perspective. One example of such a systems approach for sanitation is given by Tilley and Zurbrügg (2008) who define a system as the sum of flowstreams, which are in turn the sum of waste products (e.g. flows such as urine and faeces) and product-specific technologies. Processes in a sanitation system include, for example, user interfaces (e.g. toilets), transport, treatment, disposal and reuse. A process, that used to be closely related to sanitation in the past in Europe and still is in many other regions of the world, is agriculture, which can serve as destination for reuse. In this thesis it is argued that recognition of agriculture is indispensable for a thorough sanitation systems analysis. The purpose of a system analysis is often to support environmental management and give guidance on choosing between alternative courses of action, i.e. provide decision support. Different methods such as mathematical models and optimisation algorithms exist for decision support in sanitation management.

The focus of this thesis is not on decision-support systems or expert systems, as developed, for example, by Balkema (2003), Feng (2009), Loetscher (2000) and Finney and Gearhart (2004), but on the assessment of specific sanitation alternatives. An analysis of particular systems has the advantage that the systems under consideration can be easily adapted to a specific situation, without relying on the quality of the knowledge fed into the computer modules of expert systems. On the other hand, this presupposes that the user has certain level of knowledge of system elements used (i.e. flowstreams and processes).

One of the most inclusive and indisputable objectives of systems analysis is assessing the sustainability of a given system. The Brundlandt report in 1987 (WCED, 1987) and the UN conference in Rio de Janeiro in 1992 popularised the concept of sustainable development. Usually, the concept includes environmental, economic and social dimensions[2]. Yet, the definition of precise criteria is difficult and the meaning of the term is sometimes ambiguous since it is a rather abstract concept and it can be used in a variety of contexts. Different interpretations of sustainability exist (Guio-Torres, 2006) and sustainability criteria may even vary across time and space. A generic definition of sustainability with regard to sanitation is given by the Sustainable Sanitation Alliance (SuSanA, 2009):

> "The main objective of a sanitation system is to protect and promote human health by providing a clean environment and breaking the cycle of disease. In order to be sustainable, a sanitation system has to be not only economically viable, socially acceptable, and technically and institutionally appropriate, it should also protect the environment and the natural resources."

Several authors have compiled criteria that are relevant for sustainable water and wastewater management. One of the most extensive lists is provided by Kvarnström and af Petersen (2004), including the five categories: health, environment, economy, socio-culture and technical function. Forty-one criteria are listed in these five categories, showing the high complexity of measuring sustainability. The criteria can be useful for ranking different sanitation options. However, they do not measure the absolute sustainability of a system. That means that no targets and reference values can be scientifically set. Some even argue that there is probably no system that is absolutely sustainable and that "the concept of sustainability is more of a direction rather than a stage to reach" (SuSanA, 2009). For local decision making a weighting of the criteria might be needed to reflect the specific framework and priorities.

With the concept of sustainability in mind, the Bellagio principles for sustainable sanitation were developed and the Water Supply and Sanitation Collaborative Council endorsed them in 2000. These principles include aspects such as human dignity, quality of life, environmental security and stakeholder participation. In addition it is stated that waste should be considered as resource and that environmental sanitation problems should be resolved in domains with minimum practical size starting at households and communities.

The selection of appropriate system boundaries is another important issue to consider for sustainability assessments. An underlying principle of sustainability requires that

[2] Sometimes also an institutional or cultural dimension is included.

the export of problems in either space or time be avoided. Lundin (2003) mentions possible system boundaries for sustainability assessments as being process-defined (e.g. a wastewater treatment plant), company-defined (e.g. a wastewater utility) or extended including surrounding systems. Due to the interaction of the different processes in urban sanitation systems, it is argued in this study that an extended system boundary (see above) is essential for a complete assessment. Not only single technologies should be evaluated, but holistic strategies need to be developed. A technology-oriented or process-oriented perspective often results in only small adaptations without questioning the suitability of the overall approach. The aim needs to be to not only solve one problem at a time, but to shift to conceptual thinking to find strategies to develop and improve entire systems. Therefore, this study follows the approach of a system analysis.

2.2 Criteria for resource efficiency assessments of sanitation systems

The scope of this present study is on the assessment of the physical (i.e. environmental) dimension of sustainability. This includes environmental impacts as well as use of natural resources. For the case of sanitation, the Sustainable Sanitation Alliance defines the physical dimension of sustainability as follows (SuSanA, 2009):

> It "involves the required energy, water and other natural resources for construction, operation and maintenance of the system, as well as the potential emissions to the environment resulting from its use. It also includes the degree of recycling and reuse practiced and the effects of these (e.g. reusing wastewater; returning nutrients and organic material to agriculture), and the protection of other non-renewable resources, e.g. through the production of renewable energies (such as biogas)."

The working group on new sanitation concepts of the German Water Association (DWA) includes the following criteria into the objective "protection of environment and natural resources" (DWA (ed.), 2008):

- emissions of nutrients, oxygen depleting substances and suspended matter to water sources
- emissions of ecotoxic substances to water sources and soil
- emissions of greenhouse gases (CO_2, CH_4, N_2O)
- resource recovery (recovery of N and P, use of biogas, recycling of water)
- use of (non-renewable) resources (use of energy and material for operation and construction of sanitation systems, use of land)

The last two criteria fall in the category resource efficiency. This term refers to a combined environmental and economic strategy aiming at an economical and optimal use of resources. Resource efficiency means to achieve greater output, i.e. a product or service, with the same input or alternatively to achieve the same output with lower input. Resources can be classified as to whether they are renewable or not, whether they are essential or substitutable, and how long the natural stock is going to last. Of particular significance are resources that are finite and essential, such as phosphorus[3] (Dockhorn, 2007).

The concept of eco-efficiency is very closely related to resource efficiency and sometimes even used synonymously. Yet, eco-efficiency goes slightly beyond the use of resources by including environmental impacts associated with a product or service. The concepts of resource efficiency and eco-efficiency were primarily taken up by companies to assess their performance, implement Agenda 21[(4)] and identify advantages for their processes. Overall, the terms refer to a management concept aimed at increasing sustainability. This concept is also gradually finding its way into strategic urban planning (Reutter, 2007).

This study focuses mainly on the criteria resource recovery and the use of energy. In addition, total costs are included since economic aspects complement the concept of efficiency. Furthermore, emissions of nutrients and organic matter into water sources are assessed, because these parameters represent the conventional concept of efficiency in wastewater treatment. Use of additional physical resources such as materials and land use are not included within the framework of this study. Although there might be considerable differences among alternative sanitation concepts, the overall impact of these resources is considered to be rather marginal. Remy and Ruhland (2006) report that source separating systems with multiple piping networks need more energy for construction, but that in general the environmental impacts due to the construction phase amount to only 1.5-4% of the total impacts. Therefore, the construction phase is neglected in this study. Also greenhouse gas emissions such as CO_2, CH_4 and N_2O are not included in the assessment, but are to some extent accounted for by overall energy consumption[5].

[3] Crude oil, by contrast, is also a finite resource, but substitutable. According to Dockhorn (2007) such considerations are not (yet) reflected in market prices, but scarcity is expected to be a factor leading to price increases in future (see also Section 2.4.1).

[4] Agenda 21 is an initiative by the United Nations toward sustainable development, requiring actions on a global, national and local level.

[5] In this context it should be noted that alternative sanitation systems producing biogas have the potential to generate energy from renewable sources, and therefore reduce greenhouse gas emissions. If however, the biogas is not fully used, or if, for example, methane escapes via the effluent, this can contribute to greenhouse gas emissions.

2.3 Water

Water is one of the most important resources, since clean and sufficient water is essential for human life. Water undergoes a natural hydrologic cycle of precipitation, runoff and evapotranspiration and can therefore be considered a renewable resource. Nevertheless, degradation of water quality and water availability is of major concern on a local and regional level. Water scarcity is a potential barrier for economic development and can result in migration, social deprivation and possibly even in violent conflicts (Fröhlich and Ratsch, 2005). Often, water scarcity is a result of human maladministration and lack of finances. This is highlighted by the difference between physical water scarcity and economic water scarcity shown in Figure 2.1.

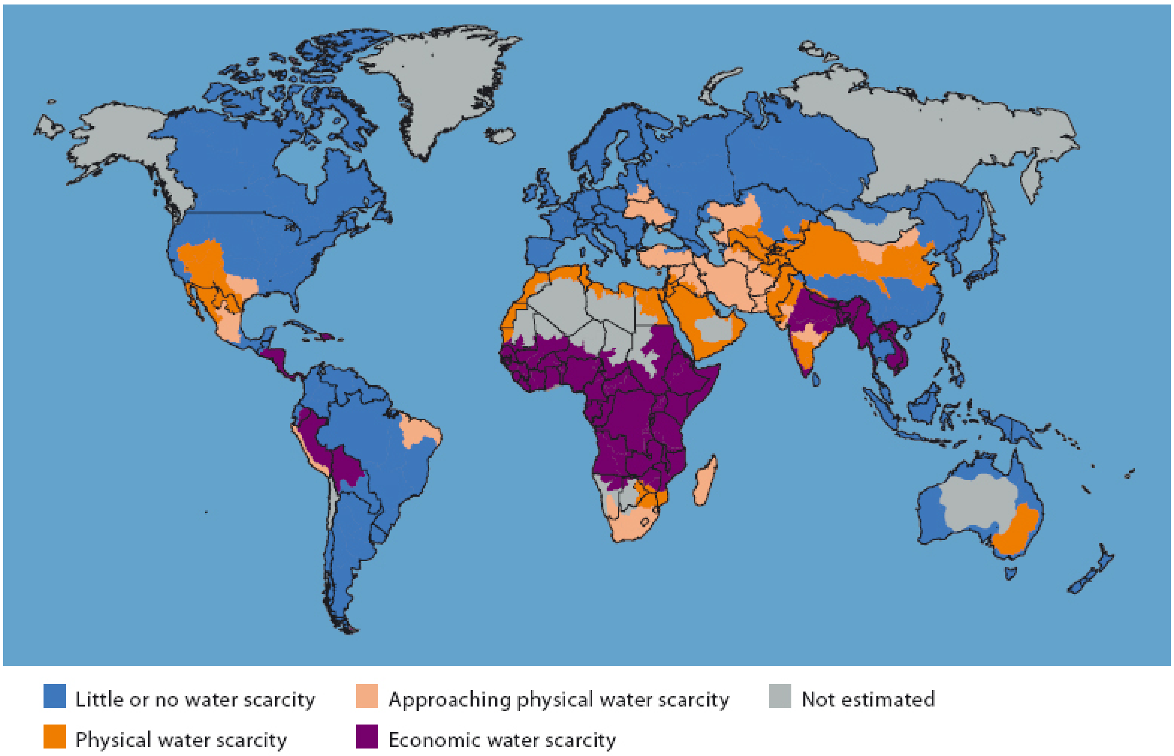

Figure 2.1: Global physical and economic water scarcity (IWMI, 2007)

Lack of clean water directly affects health, education and income. Water protection is therefore an important aspect of sustained provision of resources. The European Commission (2002) states that about 20% of all surface water in the European Union is seriously threatened by pollution. The report further warns that 60% of the European cities overexploit their groundwater resources. Globally, the provision of safe drinking water is one of the greatest challenges for sustainable development; this issue has been included in the UN Millennium Development Goals (MDG). The achievements of the MDG campaign are reflected by the fact that drinking water coverage increased worldwide from 77% in 1990 to 87% in 2006 (WHO and UNICEF, 2008). Yet, access to improved sources as represented by this statistic does not reveal any information about water safety and general availability of water, particularly in the light of population growth and increasing demand.

Water and sanitation cannot be considered separately, but are linked in many ways. First, water plays a major role for personal hygiene. Second, water is also related to communal hygiene when it is used as a means of transport for human excreta, which is often criticised as being wasteful. Last but not least, inappropriate sanitation and wastewater management affects water quality by polluting groundwater and surface water resources.

The following sections provide an overview of water use in Germany and Ethiopia and elaborate on the aspect of reusing water.

2.3.1 Water use and wastewater generation

Agriculture is the main consumer and also the main polluter of water on a global level. In developing countries agriculture can require up to 90% of the national water demand (Brugger, 2005)[6]. In this context also the concept of virtual water needs to be mentioned, which refers to the sum of water use in the production chain of a product. This implies that the actual water use (i.e. global water use) of a country can be a lot higher due to imports. The focus of this thesis, though, is on domestic water use[7].

Germany

Natural water resources (groundwater, surface water and spring water) are plentiful in Germany. On average $188*10^9\ m^3\ y^{-1}$ is available, of which 19% is used for purposes such as cooling of power plants, mining, as well as municipal water supply. Agriculture amounts to only 0.45% of the water used (UBA, 2007b). The largest share of total water consumption is for energy supply (cooling of thermal plants, about 58%) (Destatis, 2007). Municipal water supply uses only 3% of the water resources (bdew, 2008).

Germany is currently one of the industrial countries with the lowest per capita water consumption (bdew, 2008). The overall water consumption including industry and trade amounts to about $153\ l\ p^{-1}\ d^{-1}$ (Statistisches Bundesamt, 2009). Despite the

[6] Meat consumption has a significant impact on water consumption in agriculture. According to Zehnder et al. (2003) (cited in Brugger, 2005) a ratio of 20% meat consumption increases the water demand by four times compared to a vegetarian diet. In addition to nutritional habits, the efficiency of applied irrigation techniques largely affect the water requirements of agriculture.

[7] Interested readers are referred to literature, e.g. Water Footprint Network (2009) and FAO (2009a).

projection of increasing water use in the past[8] domestic water consumption has decreased constantly since the 1990s. Considering an average domestic water consumption of 147 l p^{-1} d^{-1} in 1990, the amount has dropped significantly to 124 l p^{-1} d^{-1} in 2007 (bdew, 2008; ATT et al., 2008). There are also regional variations, which are highlighted by comparing the specific water consumption in the new federal states of Germany, which is less than 93 l p^{-1} d^{-1}, with the per capita water consumption in the old federal states of about 125 l p^{-1} d^{-1} (Statistisches Bundesamt, 2009)[9]. The decrease in water consumption, which has been unanticipated by many water engineers and officials, is a result of increasing environmental awareness and the desire for economic savings by water savings.

Households consume water for various purposes. Only a small part, namely 4-5 l p^{-1} d^{-1}, is for drinking and cooking purposes (UBA, 2007a). Consumption of bottled drinking water, which is considerably more costly and energy intensive, averages 0.35 l p^{-1} d^{-1} in 2005[(10)] (Worldwatch Institute, 2007). The largest part of water consumption is for purposes such as washing, cleaning, bathing and showering; the wastewater from these activities is commonly referred to as greywater. Greywater volume is greatly dependent on household habits as well as appliances used in the households. A literature analysis of 21 European references (see Annex B) revealed an average greywater volume of 105 l p^{-1} d^{-1}. Yet, the four German references give an average of only 85 l p^{-1} d^{-1}, which is expected to further decrease in future[11]. Also water consumption for toilet flushing has decreased significantly in the last decades. A change in technology has resulted in reduced volumes required for flushing. Dual or adaptable flush systems that allow the reduction of the flushing volume are becoming more and more common. Thus, it is difficult to establish average values and a high variability exists. Table 2.1 summarises values derived from the literature and illustrates the average values that are used in the

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[8] Forecasts in the 1970s for the old federal states predicted an increasing water demand to more than 200 litres per person and day (Schleich and Hillenbrand, 2007).

[9] The main reasons for these variations are differences in prices and income. In addition, the rate of diffusion of water saving appliances and technologies and the motivation for water savings is a lot higher in the new federal states due to modernisations of the housing stock and increased economic awareness (Schleich and Hillenbrand, 2007).

[10] Regarding the total consumed volume of bottled water Germany ranks 6th worldwide with more than 10,000 million litres being sold in 2005. Considering the specific water consumption per person only the inhabitants of Italy, Spain, France and Mexico consume more bottled water per year (Source: International Bottled Water Association cited in Worldwatch Institute, 2007).

[11] For example, in the ecological settlement Lübeck-Flintenbreite the long-term average greywater volume is 61 l p^{-1} d^{-1} (Oldenburg et al., 2008a).

mass flow model in this study for the case of Hamburg (see Section 4). The averages are based on the household water consumption in Hamburg of 107 l p^{-1} d^{-1} (Hamburg Wasser, 2007).

Table 2.1: Specific water consumption for different household activities in Germany

	selected average l p^{-1} d^{-1}	minimum - maximum l p^{-1} d^{-1}
Greywater		
Personal hygiene	41	30-55
Laundry	15	10-40
Dishes	8	4-10
Cleaning, gardening, car washing[12]	8	3-10
Eating, drinking	5	3-6
Bottled water	0.35	0.2-0.5
Toilet flushing		
Conventional toilet	30	20-50
Vacuum toilet	6	3-9
UDDT, urine flush	0	0-1.2
UDDT, faeces flush	3	0.5-6
Dry toilets	0	0
Small trade	11	

Values based on UBA (2007a); Fehr (2007); Messerschmidt (2008); Hiessl et al. (2003); Balkema (2003); Hess. Umweltministerium (1994) cited in Lange and Otterpohl (2000); Leonhardt (2005); Worldwatch Institute (2007)

Ethiopia

Water availability in Ethiopia is characterised by regional and seasonal differences. With an average precipitation rate of 850 mm y^{-1} the total actual renewable water resources amount to about $122*10^9$ m^3 y^{-1} (FAO and Kundell, 2009). Yet, overall availability and quality of the supplies is considered to be one of the lowest of the world (IRIN, 2003). In particular a lack of storage capacities and institutional capacity is the main reason for this dilemma. Water withdrawal is about 5% of the available resources, of which 94% are for agricultural use, 5% for domestic use and 1% for industrial use (FAO and Kundell, 2009). Water quality is a major concern. Recent outbreaks of water-related epidemics such as cholera and an estimated 15% of the deaths in Ethiopia are attributed to water or wastewater (Stedman, 2008).

[12] Half of this volume refers to water that is "lost" in the system, e.g. water that evaporates and cannot be recycled as process water any more. Water from cleaning is usually disposed of in the drainage and can thus be recycled.

Water supply coverage increased from 13% in 1990 to 42% in 2006. However, there are large discrepancies between rural areas, where coverage is about 31%, and urban areas, where coverage is about 96% (WHO and UNICEF, 2008). The prevailing type of water supply is public stand pipes, wells or boreholes, while only 9% of the total population have access to house connections (in urban areas this ratio is 50%) (WHO and UNICEF, 2008). The household water consumption differs according to the type of water supply and the household size. In a study on 40 households in Arba Minch carried out by Teklemariam (2009), water consumption per capita was up to 55 l p^{-1} d^{-1}, but the majority used only 4 to 20 l p^{-1} d^{-1}. The demand assessment and baseline study of the ROSA project gives an average water consumption of about 40 l p^{-1} d^{-1} if the household has a house connection (AMU and ARB, 2007). If however water from standpipes is used, the consumption per capita is a lot lower. In Arba Minch about 22% of the households use water from these sources. For that case, the per capita consumption can be calculated to be about 5 l p^{-1} d^{-1} based on data from AMU and ARB (2007).

The Ethiopian Ministry of Water Resources provides estimates of water demand for different household activities (see Table 2.2). Comparing the total demand with actual consumption values, as discussed for the case of Arba Minch, shows that only the demand for yard connections seems to tie in with the corresponding figure given in Table 2.2. Water consumption from public standpipes is most probably a lot lower than the indicated 30 l p^{-1} d^{-1}. Average values used for the assessment of the case study Arba Minch are given in section 5.1.3. Bottled water is available in Ethiopia, but there is no official data about average consumption, which is estimated to be rather low. Therefore, bottled water is neglected in the assessment. Nevertheless, the issue might become important if the dissemination continues and might have to be investigated in further studies.

Sewer systems exist only in five towns in Ethiopia and even there serve only a minority of the population[13]. Thus, the wastewater is usually disposed of in backyards or onto streets. Since water-flushed toilets are used only by a fraction of the population, human excreta and liquid wastes such as greywater are usually separated. Occasionally greywater and blackwater are mixed, e.g. when greywater is disposed of in pit latrines and septic tanks.

[13] The sewerage and pond system for wastewater treatment in Addis Ababa is designed to serve only 5% of the town's population (MoW et al., 2007).

Table 2.2: Specific water consumption for household activities in Ethiopia depending on the type of water supply (based on Ministry of Water Resources, 1997 cited in AMU and ARB, 2007)

	House connection $l\ p^{-1}\ d^{-1}$	Yard connection $l\ p^{-1}\ d^{-1}$	Public standpipe $l\ p^{-1}\ d^{-1}$
Eating, drinking	10	8	7
Personal hygiene	37	16	10
Laundry	15	8	7
Dishes	5	4	4
Cleaning	7	3	2
Toilet	6	1	
Total	80	40	30

2.3.2 Water reuse

Reusing treated wastewater can contribute to a more efficient use of available water resources. Source separation of greywater has a favourable effect because greywater is a comparatively less concentrated wastewater flow and the treatment processes can be adapted accordingly to make the water fit for reuse. Reusing greywater compared to reusing mixed wastewater additionally seems to achieve social acceptance more easily. Occasionally, however, high levels of indicators for faecal contamination can also be found in greywater (Eriksson et al., 2002). In addition, it should be noted that greywater is highly variable in its composition, depending on the prior use of the water. Studies from developing countries have shown that greywater can have much higher pollutant concentrations than usually reported due to lower water consumption and multiple uses of the water (Imhof et al., 2005; Teklemariam, 2009). Recycling of greywater is particularly suited to decentralised reuse schemes, whereas, up to now, mixed wastewater is mostly treated for reuse in centralised schemes. The level of treatment also interlinks with the purpose of the reuse. Decentralised reclamation usually allows the direct reuse for household purposes, such as toilet flushing, but also for other purposes such as laundry or dishes. Centralised reclamation schemes often provide irrigation water for agriculture, but also process water to households, which are supplied by a dual reticulation system. Of course other uses such as industrial use or groundwater recharge are possible. In addition, it is technically feasible to provide water for human consumption on a decentralised, as well as on a centralised level. Although maybe not yet widely accepted by the general public (and even prohibited in some countries), this is already implemented in several case studies (see below) (Salgot et al., 2006). In developing countries, the main use of reclaimed water is irrigation, since the reliability of treatment processes required for more advanced reclamation uses is insufficient, e.g. due to power cuts.

The level of treatment and the kind of technology applied depends on the use of the reclaimed water. In principal, anaerobic as well as aerobic treatment is suitable as biological treatment. Treatment can also be done on a decentralised level down at household, particularly since membrane filtration has conquered the market. The pore size of membranes determines their filtrating effectiveness. The smaller the pore size, the higher the energy requirements for providing the pressure and the higher the chemical requirements for cleaning the membranes. Currently, a wide range of research and other activities are established with regard to membranes in wastewater and greywater treatment[14] Usually, a multi-barrier concept is applied for reuse treatment. Further disinfection for advanced reuse can be achieved by UV or ozone (Boller, 2006; Knerr et al., 2008).

In economically less developed countries, such as Ethiopia, other technologies, which are more low-cost and more robust, need to be implemented for reusing greywater or wastewater. For example, on a centralised level wastewater treated in extensive pond systems can be reused for agricultural purposes (Neubert, 2003). On a decentralised level, greywater towers or planted soil filters represent a way of reusing greywater (Ayele, 2009). It is important to differentiate between controlled reuse and the most often unplanned use of wastewater as irrigation water as it is practiced world-wide in developing countries. Stedman (2007) estimates that about one tenth of the world's crops are irrigated with wastewater. A study of 53 towns in Africa, Asia and Latin America concluded that in 80% of the towns non- or partly-treated wastewater is used as irrigation water, and in 70% of the towns more than half of urban agriculture is dependent on wastewater (Haskins and Dold, 2008). Therefore, it is crucial to acknowledge the importance of wastewater as water source and include proper treatment for safe reuse.

Worldwide, many examples of greywater and wastewater reclamation schemes for various purposes exist[15]. In Germany, particularly decentralised schemes using greywater reclamation are implemented. According to Nolde (2005) up to 2005 there were already 300-400 greywater recycling systems installed in Germany. In Hamburg, greywater recycling is promoted by financial incentives and the waste department has implemented a large scale installation in 2005 (Pontos GmbH, 2006). In Germany there

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[14] See for example, Gnirss et al. (2003), Li et al. (2008), Kubin (2004) and Nolde (2005).

[15] Reclamation of mixed wastewater for (indirect) potable uses is practised, for example, in Windhoek, Namibia (Haarhoff and van der Merwe, 1996), in Singapore (NEWater plants) (Tao et al., 2006) and in areas with water shortages in Australia (Traves et al., 2008). In some cases dual reticulation, i.e. one pipe for drinking water and one pipe for reclaimed water, is in place, including prominent examples such as the Olympic Park in Sydney and the Millennium Dome in London (Stedman, 2007).

is however a degree of controversy in discussions about water saving, and thus also the recycling of wastewater. Water supply and sewer systems were often designed on the assumption of increasing water demand. In addition, fee collection is connected to actual water use and critics argue that unit costs, i.e. euro per cubic metre water, are going to increase if water consumption decreases (Leist and Magoulas, 2002). Despite the possible increase in specific costs, planning mistakes should not be passed onto the population, and thus the potential of water reuse needs to be considered also from a holistic perspective with respect to available resources. The reclamation of grey- or wastewater is favourable compared to rainwater use, since its availability is directly related to consumption, therefore requiring smaller storage facilities (Zhang et al., 2009). All in all wastewater reclamation impacts not only on water supply, but also on wastewater treatment. Potential advantages and barriers should be considered from a holistic planning perspective to use the resource *water* in the best way possible.

2.4 Nutrients

Nutrients are essential for human and plant nourishment. The importance of nutrients and fertilisers is presented in the next section, followed by a discussion of the connection between nutrients and wastewater. Furthermore, nutrient recycling from mixed wastewater and from source separated wastewater flows is introduced; this includes the discussion of selected technologies, which are used in the system analyses in sections 4 and 5.

2.4.1 Nutrients and fertilisers

A variety of chemical substances are classified as nutrients and differentiation is usually made between macronutrients, which are required in larger amounts, and micronutrients, which are required in relatively small quantities. The former include nitrogen (N), phosphorus (P), potassium (K), sulphur (S), all of which are dealt with in this study, as well as calcium (Ca) and magnesium (Mg).

Harvesting of crops extracts nutrients from cultivated land. Therefore, sustainable agriculture depends on the addition of nutrients to sustain the fertility of soils through external sources, i.e. the supply of fertilisers. According to the Law of the Minimum developed by Justus Liebig, agricultural growth is controlled by the availability of the scarcest resource. Thus, the scarcest nutrient limits the plant growth. But the excess application of nutrients can also be harmful to plants and additionally result in environmental problems, e.g. by leaching. Therefore, balanced fertilisation, including an appropriate dosage of all nutrients, is essential for plant growth. Nutrient requirements are crop specific, and also depend on soil type, climate, irrigation and other factors. Thus, all these factors need to be taken into account for optimum nutrient supply. In general, the specific diet, e.g. the ratio of meat and dairy products, impacts on the overall amount of fertiliser needed. According to Schmid Neset (2005), increased human meat consumption in Sweden over the past 130 years has increased the demand for phosphorus by about 25%, and nitrogen by about 15%.

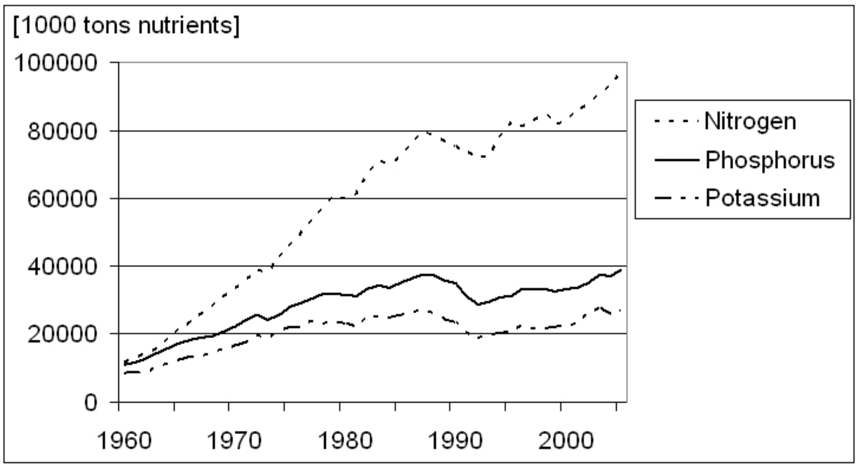

Figure 2.2: Global fertiliser consumption (Data based on IFA, 2009)

Fertilisers fall into one of two categories; natural fertiliser which are organic, or manufactured fertilisers (also called mineral fertiliser) which are inorganic. The former includes, for example, decayed plant or animal matter such as compost, worm casting or manure. Organic fertilisers have been applied for more than 2000 years providing not only valuable nutrients, but also organic matter to the soil, thus improving overall soil conditions (Moss et al., 2002). Perceived disadvantages of organic fertilisers include their variable nutrient content and their relatively large volume affecting transport and application. After the industrial development of mineral fertilisers in the early 20th century, and the subsequent Green Revolution era after World War II, the fertiliser use worldwide changed drastically. Figure 2.2 illustrates the significant increase in N, P and K fertiliser consumption in the last decades.

Fertiliser consumption differs significantly across continents as is illustrated in Figure 2.3. Worldwide, nutrients are not supplied in sufficient amounts. For example, Liu et al. (2008) estimated that the net loss of phosphorus from the world's cropland due to insufficient supply, runoff and erosion, amounts to more than 10 million tons every year. Particularly in Africa soil degradation and loss of fertility impacts greatly on crop yields and aggravates poverty and malnourishment. UNEP and Monosson (2008) estimate that about 40 million hectares of land in Africa are nutrient-deficient[16]. In Germany, N, P, K and S fertilisers are applied according to recommendations based on

[16] One reason for this is overpricing of mineral fertilisers. According to Millenium Project (2006) African farmers pay two to six times more than the world market prices for mineral fertilisers.

soil nutrient content. Average nutrient application rates are shown in Section 4.1.1. In Ethiopia, only N and P are available as mineral fertilisers and application rates are usually significantly lower than recommended. Total consumption of N and P fertilisers in Ethiopia in 2006 amounted to 98,000 t N and 84,000 t P_2O_5, compared to 1,600,000 t N and 265,000 t P_2O_5 in Germany (IFA, 2009). This discrepancy is even more striking when one considers that the total agricultural area in Ethiopia is about twice that of Germany (based on data from World Bank, 2009b). According to Demilew (2008) even in those regions with high fertiliser consumption, only three out of four farmers apply fertiliser and their rate of application is usually less than a third of the recommended rate (see also Section 5.1.1 for the case of Arba Minch).

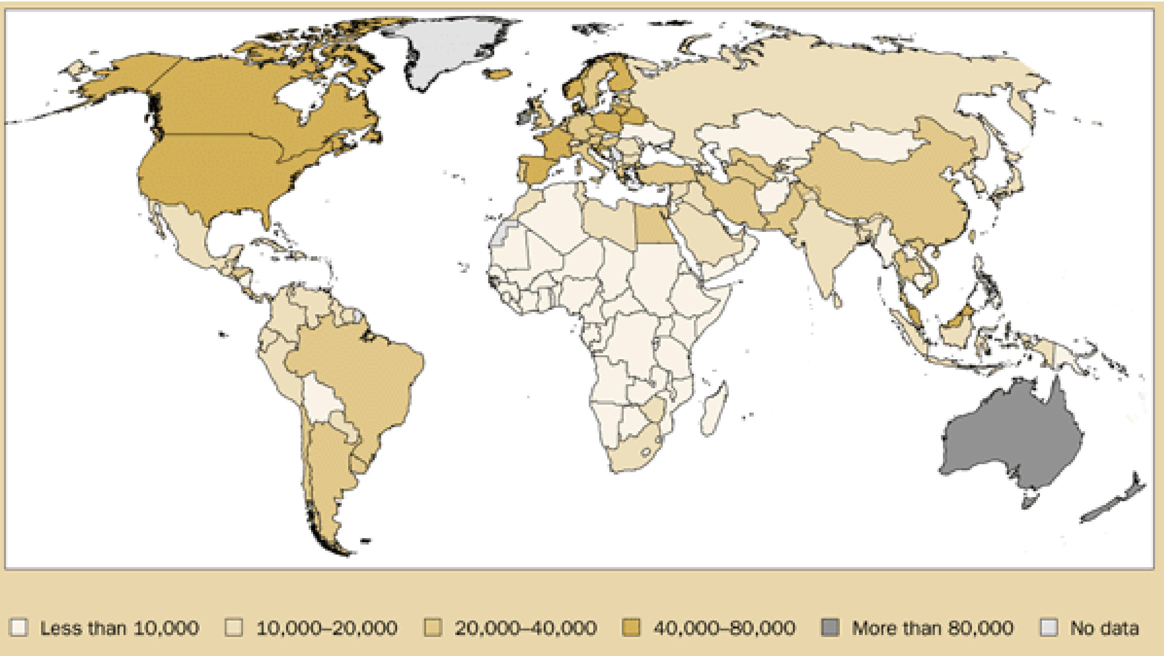

Figure 2.3: Distribution of global fertiliser consumption in 2001 [in tons per million people] (Source: World Bank cited in MilleniumProject, 2006)

One of the problems related to fertiliser use is the risk of over-fertilisation and loss of nutrients in runoff, erosion or infiltration (Behrendt et al., 2003). This can result in eutrophication of the receiving waters and other environmental impacts such as nitrate pollution of drinking water sources. According to Jayan (2004), about 32% of applied fertilisers reach surface waters and groundwater.

Another area of concern is that mineral fertiliser production and transport is energy intensive and entails other environmental impacts such as greenhouse gas emissions and water pollution[17]. In 1995, energy consumption for fertiliser production amounted

[17] For example, the German fertiliser company K+S pollutes surface waters by discharging annually 7 million m^3 untreated brine from potassium mining (Quasthoff, 2009).

to 1.2% of energy consumed world-wide[18]. In addition, fertiliser production results in gaseous emissions of e.g. CO_2, NO_x and N_2O. For example, N_2O emissions attributed to fertiliser production amount to about 8.5% of total N_2O emissions (Patyk and Reinhardt, 1997). Furthermore, critics argue that mineral fertilisers often contain traces of uranium, cadmium and other heavy metals, which accumulate on fertilised land, build up in produce, and end up in humans (Kratz, 2004; Schuh, 2005a). The different standard that is applied to mineral fertilisers as compared to other fertilisers is of particular concern here. For example, despite the fact that EU regulations on heavy metals already exist for secondary fertilisers, EU standards for mineral fertilisers are only going to be effective by 2017.

Contamination of raw fertiliser resources (particularly phosphate) is dependent on their origin[19]. It should be noted in this context that phosphorus reserves show a particularly skewed geographical distribution. Commercially viable reserves are available in only a few countries. Phosphate rock mining amounted to 167 mio t y^{-1} in 2008 of which 17% were mined in Morocco and Western Sahara, 19% in the USA and 30% in China (USGS, 2009). China uses most of its phosphate domestically, so that global P reserves are subject to what Rosemarin (2004) calls "precarious geopolitics". Also potassium is produced only in about a dozen countries including Canada, the Russian Federation, Belarus and Germany (Ober, 2007). Phosphorus and potassium are both limited resources and particularly phosphorus reserves are considered to be scarce. Cordell et al. (2009) calculated the expected global peak in phosphorus production to occur around 2030. Assuming current mining rates, the phosphorus reserves of 15,000 mio t would be sufficient for about 90 years (USGS, 2009). Additionally, phosphorus consumption and, thus, mining rates are expected to increase, resulting in an accelerated decrease of the reserves. On the other hand, reserve bases of an estimated 47,000 mio t, which include those reserves that are currently economically unviable, could in future be used for phosphate rock mining. Yet, this will most probably coincide with high price increases since the quality of reserve bases is usually inferior and mining of these ores is more expensive.

Recent fertiliser price increases in 2007 and 2008 indicated how the market might develop. Although fertiliser prices on the world market declined again in 2009, farmers were hit hard by skyrocketing prices with increases of up to more than 300% (World Bank, 2009a; Demilew, 2008). Particularly farmers in Africa, who need the fertilisers to

[18] Nitrogen fertiliser alone contributed to 0.94% of total energy use (Patyk and Reinhardt, 1997).

[19] Sedimentary reserves show a particular high heavy metal contamination compared to igneous reserves (Kratz, 2004). Segregation of heavy metals is possible, but cost-intensive.

replenish the nutrient-depleted soil, struggled with the crisis caused by high energy and transport prices as well as increased demand for fertilisers as a result of biofuel production (Mongabay, 2008). The following factors are expected to impact on fertiliser availability and fertiliser prices in the long-term:

- Population increase and associated increase in food demand. In addition, food demand increases as a result of efforts to eradicate undernourishment.
- Change of dietary patterns leading to increased meat consumption[20] particularly in Asian and African countries and therefore increased nutrient demand for fodder crops.
- Expansion of bio-fuel production[21].
- Decrease in arable soil (due to e.g. soil erosion, urbanisation and industrial development) and imponderability in farming (e.g. due to climate change) requiring increase in productivity.
- Poor soils and current under-utilisation of fertiliser particularly in Africa.
- Finiteness of phosphorus and potassium reserves and their availability in only a few countries.
- Contamination of mineral reserves with heavy metals.
- Dependency of fertiliser prices on energy prices, which are volatile.

The discussion on mineral fertilisers shows that other sources of nutrients need to be considered to decrease dependency on future driving forces and develop long-term strategies. Beside farm fertilisers (e.g. manure), which are beyond the scope of this study, waste such as organic waste and wastewater is a potentially indefinite source of nutrients. The latter one is the topic of the next section.

2.4.2 Nutrients in wastewater flows

Nutrient uptake is essential for human living, yet, almost all the nutrients that are taken up are excreted in urine or faeces[22]. In addition, detergents, personal care products, foodstuff, etc. are sources of nutrient discharges into wastewater. Nutrients in

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[20] Income growth, urbanisation and shift in consumer preferences result in increased meat demand, which is expected to rise by more than 50% by 2030 (FAO, 2008).

[21] Estimations of anticipated fertiliser consumption for bio-fuel production are variable but go as high as 28% in 2011 (Cassman et al., 2006 cited in FAO, 2008).

[22] Only children accumulate a few percent of the nutrients consumed in their body (Jönsson and Vinneras, 2004).

wastewater are considered in this study from two perspectives. First, nitrogen and phosphorus can lead to pollution of water sources. Therefore, emissions e.g. from combined sewer overflows or from wastewater treatment plant outlets, need to be avoided. In this context, it needs to be kept in mind that nutrient elimination is one of the most energy-consuming steps in wastewater treatment. Second, nutrients in wastewater represent a potential source of nutrient supply for agricultural purposes and can therefore contribute to mineral fertiliser replacement.

The nutrient content of wastewater flows is rather variable. For example, the nutrient load in excreta and the allocation to the respective flows, i.e. faeces and urine, depend on the kind of nutrition and the digestibility of the food. Furthermore, most data on nutrients in wastewater are only available for mixed wastewater and not for source separated flows. Literature reviews on nutrient composition of source separated flows were done by e.g. Henze (1997), Herrmann and Klaus (1997), Londong and Hartmann (2006) as well as Niederste-Hollenberg and Otterpohl (2000). Each of these reviews however, are based on a maximum number of ten values per parameter and all authors conclude that the database needs to be increased in order to eliminate the effect of dissimilarities. Therefore, this study uses average values derived by a review done by the author for the task group on new sanitation concepts of the German Water Association (DWA) (Meinzinger and Oldenburg, 2009; see also Oldenburg et al., 2008b). The review included more than 200 European references on specific nutrient contents of wastewater and organic waste flows. The results of this review are shown in Figure 2.4 and Annex B.

The highest ratio of the nutrients N, P and K can be found in urine. Up to 80% of the daily load of nitrogen and about 50% of phosphorus and potassium are present in urine (see Figure 2.4). This suggests various benefits regarding the separation of this flow. On the one hand, urine separation can be useful in terms of reduced nutrient loads to the wastewater treatment plant or for selective inflow to the treatment plant to reduce nutrient peak flows. On the other hand, separated urine represents a flow particularly suitable for the provision of plant nutrients. Other than phosphorus, faeces contain relatively low nutrient loads. Nevertheless, this flow and its separation from the total wastewater flow can play an important role in terms of energy provision due to its high content of organic matter (see Section 2.5) and can be considered for provision of plant nutrients in combination with urine. Greywater contains relatively high loads of sulphur, but also nitrogen, phosphorus and potassium are present. Greywater characteristics are highly dependent on the habits of the users, appliances in the household (e.g. dishwasher) and the detergents that are being used. This impact can be observed, for example, by looking at phosphorus contents in greywater. The last decades have seen a general decrease in average P contents in domestic wastewater in Europe due to the ban on phosphates in washing agents in many countries. Nowadays phosphates in detergents are replaced by organic phosphorus ingredients. However, in dishwashing detergents phosphates cannot be replaced yet. The increased use of dishwashing agents is offsetting the reduction due to detergents to the point where phosphorus loads in greywater are again increasing. Londong and Hartmann (2006) showed that the dishwashing agents in a household with three members can contribute to a daily load of about 0.4 g P per person.

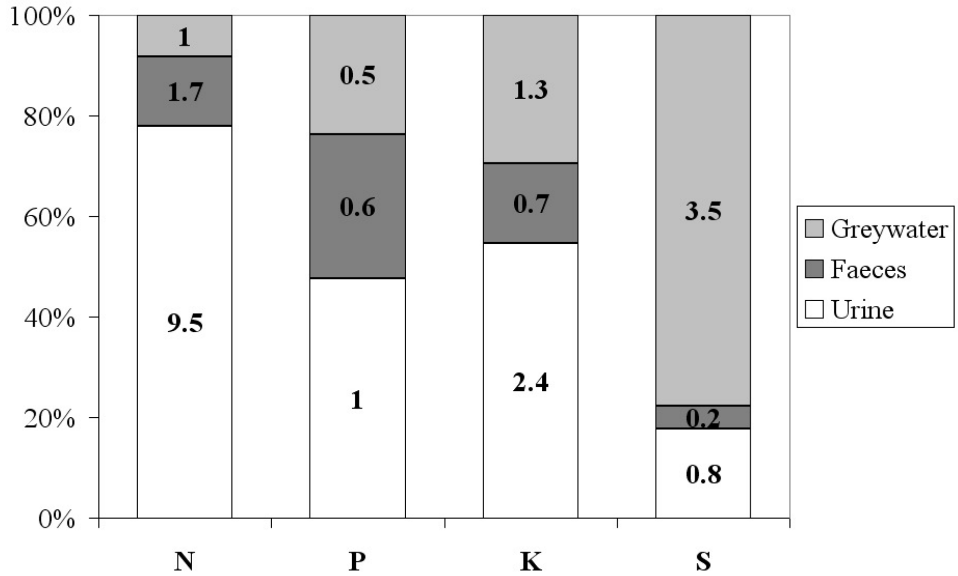

Figure 2.4: Nutrient distribution and specific loads [g p^{-1} d^{-1}] in urine, faeces and greywater (Source: compiled by the author)

The characteristic values derived above are based only on European references and reflect the nutritional habits and water-use patterns applicable to this region. Due to this dependency the context is important and for regions other than Europe representative values for such regions should be used. For regions like most of Africa, data relating to household nutrition and water usage is scarce. Jönsson et al. (2004) developed a method to calculate the expected nutrient content in excreta based on national nutritional data (i.e. food protein content), which is published by the Food and Agriculture Organization (FAO)[23]. This method is used to estimate the nutrient content in faeces and urine in the case of Arba Minch according to the following formulae (Jönsson et al., 2004)[24]:

N = 0.13 * total food protein

[23] National food balance sheets can be found on http://faostat.fao.org/site/368/default.aspx#ancor (last access on 9 September 2009).

[24] Comparing the calculated nutrient contents derived from FAO data for Germany with average data in Figure 2.4 results in differences of 16%, 3% and 39% for N, P and K respectively.

$$P = 0.011 * (\text{total food protein} + \text{vegetal food protein})$$

$$K = 0.015 * (\text{total food protein} + 4.6 * \text{vegetal food protein})$$

Based on data from FAO (2009b) on food protein content in Ethiopia[25], nitrogen, phosphorus and potassium loads are calculated to be 6.9 gN p^{-1} d^{-1}, 1.1 gP p^{-1} d^{-1} and 4.1 gK p^{-1} d^{-1} in urine and faeces together. The digestibility of the food affects the distribution of nutrients between urine and faeces. More processed and therefore easier digestible food such as consumed in the European diet, generally results in more nutrients being excreted via urine (Jönsson and Vinneras, 2004). Therefore, it can be expected that the ratio of nutrients excreted via faeces is higher in Ethiopia than in Europe. But for the purpose of this study the same distribution as derived from the European references are used. This means that 85% of N, 63% of P and 77% of K are found in urine. Also greywater characteristic values differ significantly in the African context compared to European data, since water scarcity often leads to multiple uses and therefore higher pollutant concentrations. As stated previously, for many regions in Africa, which includes Ethiopia, little data is available. In general, very high and usually very variable organic matter concentrations can be observed, e.g. with COD values reported to be as high as 8000 mg l^{-1} (Raude et al., 2009). The same study, which was carried out in a location in Kenya similar to Arba Minch in Ethiopia, reports about variable nitrogen and phosphorus concentrations ranging from 2 to 340 mg l^{-1} and 1 to 13 mg l^{-1} respectively. With a reported per capita water consumption of 10 l p^{-1} d^{-1} in this study, this translates to 0.02-3.4 gN p^{-1} d^{-1} and 0.01-0.1 gP p^{-1} d^{-1}.

Theoretically, the nutrients in excreta of the world population would be sufficient to replace roughly about a quarter of the global N fertiliser consumption, a fifth of the P fertiliser consumption, and a third of the K fertiliser consumption. Of course, this calculation is rather hypothetical, since the population distribution is not equal to the distribution of fertiliser use and it does not seem realistic to recover 100% of all nutrients from urine and faeces. These considerations also show that excreta-based nutrient content and particularly the nutrient content of single flows such as urine do not show the same proportion of nutrients as used in mineral fertiliser. Therefore, site-specific supplemental application of particular nutrients could be required in addition to the use of nutrients recovered from human waste flows.

In the next sections possibilities for nutrient recovery from mixed and source separated wastewater are introduced.

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[25] In 2003, the total food protein consumption in Ethiopia was 53.41 g p^{-1} d^{-1} and the vegetal protein consumption was 47.14 g p^{-1} d^{-1} (FAO, 2009b).

2.4.3 Nutrient recovery in conventional systems

The use of wastewater and human excreta for agricultural purposes has a long tradition. Beside the use of wastewater as irrigation water (see Section 2.3.2), farmers all over the world have known of the nutritional value of human waste for centuries. Also in Germany in the 19th century nightsoil used to be collected and transported to agricultural fields (Erismann, 1882; Salviati et al., 1865). However, the introduction of water-based sanitation and the discovery of other natural and artificial fertilisers such as guano and synthetically produced ammonia, gradually disconnected the link between sanitation and agriculture. Nowadays, the use of sewage sludge in agriculture or in composting is still common but it is clearly a declining form of nutrient recycling. Stabilised sewage sludge provides nitrogen, phosphorus and humus, whereas sewage sludge ashes only contain phosphorus[26]. However, sewage sludge is used in agriculture mainly because it is a cost effective disposal method and not because of its competitiveness as nutrient supply. While in Europe about 41% of the sewage sludge is used in agriculture, this ratio is only about 30% for Germany with a shift towards thermal utilisation (Scheidig, 2009). One important concern regarding sludge application on soil is the possible contamination with heavy metals (e.g. from industry, pipeworks, copper roofs, etc.) and organic pollutants (e.g. from personal care products and pharmaceuticals). Countries such as Switzerland and Sweden have already imposed a ban on the use of sewage sludge (Schuh, 2005b) and some German states aim at stopping this practice, too. Besides pollution by heavy metals, a low availability of P and a limited presence of other nutrients are other factors that restrict the use of sewage sludge as fertiliser replacement.

Due to rising concerns associated with the use of sewage sludge, various efforts are currently undertaken to study ways of recovering nutrients from wastewater effluent, sludge liquor, sewage sludge and sludge ash. The main focus of recovery is on phosphorus, and only a few processes also include nitrogen as one of the recovery products. The kind of wastewater treatment processes, e.g. type of phosphorus removal[27] or type of incineration process, governs the possible nutrient recovery methods that can be used. The recovery processes differ in how they make the nutrients available. For example, sludge may be dissolved using acids, bases or heat (Stark, 2002). One of the last steps usually applied for P recovery is the crystallisation or precipitation

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[26] Please note that only ashes from mono-incineration can be used. Ashes are generally subject to relatively low phosphorus availabilities and high contamination.

[27] Precipitants such as iron decrease the plant availability of P (Römer, 2006) and impact on the chemical requirements in P recovery (Hultman and Löwén, 2001).

as calcium phosphates, struvite (MAP) or potassium struvite. These recycling products were tested in crop trials to be agronomically interesting products (Römer, 2006). Most of the recovery methods[28] are currently tested at laboratory or pilot scale and only a few implementations on a larger scale exist up to now. Nitrogen is sometimes recycled at low levels by incorporation into struvite. But generally little work has been done on N recovery from wastewater due to the high dilution and the associated high effort required[29]. But in principle, technologies such as steam or air stripping are available for nitrogen recovery (Bosshart et al., 1993; Gebel et al., 1994).

Since nutrient recycling from sewage sludge and from sludge ash is also assessed within the scope of this study, two recovery processes, namely the Seaborne® process and the BioCon® process, are briefly described. They are selected for the assessment based on the availability of data from pilot scale and large scale applications.

The Seaborne® process has been tested in a pilot plant in Germany since 2000 (Seaborne EPM AG, 2003). The first full-scale application of this process at a wastewater treatment plant has been in operation since 2006 and handles a sludge volume of $110 m^3\, d^{-1}$ (Bayerle, 2009). The phosphorus in the digested sludge is made available by chemical disintegration. Addition of acids lowers the pH value and brings phosphorus, as well as heavy metals into solution. The lower the pH the more phosphorus can be made available. At pH 2 up to 95% of the phosphorus can be dissolved (Bayerle, 2009). With a pH value of 3 about 60% to 70% of phosphorus can be brought into solution. With a pH lower than 3 however, difficulties with sludge dewatering result (Montag, 2008). Heavy metals are precipitated in the next step and extracted from the system before phosphate is precipitated as MAP. Subsequently, ammonia is stripped and recovered by sulphuric acid. MAP and ammonium sulphate are nutrient products that can be used in agriculture. For a more detailed description of the Seaborne process please refer to Vesterager (2003), Müller (2005) and Bayerle (2009).

The BioCon® process was developed by the Danish company PM Energi A/S and some first experiences have been gained through applications in Sweden. Ash from sewage sludge incineration is milled and H_2SO_4 is used for dissolution before a series of ion exchangers are used for the recovery of phosphoric acid (H_3PO_4), ferric chloride (FeCl) and potassium hydrogen sulphate ($KHSO_4$). The ratio of phosphorus that is released by leaching depends on the volume of added acids, as well as the amount of metals used

[28] For an overview of promising technologies for P recovery please refer to, e.g. Balmer (2004), Levlin (2007), Montag (2008), Scheidig (2009) and von Horn (2007).

[29] According to Buer et al. (2002) nitrogen recovery by air or steam stripping is more energy-efficient, but less cost-efficient than industrial production of ammonia solution from atmospheric nitrogen.

for phosphorus precipitation (Stark, 2002). The amount of phosphorus released from sludge ash can be up to 80-90%, but in the main, the degree of phosphorus recovery of the overall process is about 60% (Balmer, 2004). An overview of the BioCon process is given by Hultman et al. (2001).

In countries with low-cost wastewater treatment technologies other options for nutrient recycling from mixed wastewater are available. As pointed out in Section 2.3.2, wastewater irrigation is a widespread practice, providing not only water, but also nutrients to agricultural areas. For example, effluents from pond treatment systems can be used in land application systems. It is however often a challenge to balance seasonal water requirements with plant uptake of nutrients (Zachritz et al., 2006). Additionally health risks are of major concern. Another method is the treatment and use of faecal sludge from sources like pit latrines or septic tanks. For example, co-composting of faecal sludge with organic waste provides a valuable product to agriculture and combines waste treatment with nutrient recovery. Drying beds can be used to dewater the faecal sludge and make it fit for composting (Cofie et al., 2006). Examples of co-composting plants include Kumasi in Ghana, Port-au-Prince in Haiti and Niono in Mali (Strauss et al., 2003). Not all the nutrients from wastewater will be available in the product and particularly nitrogen is lost as ammonia (Tanner, 2003). But the compost does provide organic matter, which improves the overall soil conditions. More information on process dynamics, technical and financial issues related to faecal sludge management and co-composting can be found in Cofie et al. (2009), Steiner et al. (2002) and Strauss et al. (2003).

2.4.4 Source separating systems

Source separation of wastewater allows a more specific treatment and easier recovery of nutrients from source separated flows. Smaller volumes and therefore higher concentrations reduce the effort required for recovery as compared to mixed wastewater. Source separated flows can also be more easily handled, and in the case of urine and greywater, provide a substrate that is safer to manage. In the following sections, the possibilities for reusing nutrients from the three source separated flows, namely urine, faeces and blackwater, are briefly discussed. An overview of international case studies on source-separating sanitation projects is given, for example, by SuSanA (2010). Innovative projects including source separation particularly in Europe are discussed by Hegger (2007).

Urine

Due to the relatively high nutrient concentration in urine (see Section 2.4.2), source separated urine represents a valuable liquid fertiliser that does actually not need any further processing. However, storage is recommended to decrease the pathogen content (WHO, 2006). Recommended storage time depends particularly on climate, e.g. ambient temperature, and also on the crops to be fertilised. Niwagaba (2009) concludes that storing urine for a few weeks at 30-34°C or up to 2 months at 20°C is sufficient to allow for unrestricted use if urine is undiluted or diluted with water at a ratio of less than 1:1 (water:urine). Several studies have shown that the fertiliser effect of urine particularly with regard to nitrogen and phosphorus is comparable to mineral fertiliser (e.g. Kirchmann and Petterson, 1995; Muskolus, 2008). In order to reduce the risk of nitrogen volatilisation, urine should be stored in covered containers and spreading should be done cautiously. In this case, NH_4-N losses of about 5% during storage and 5% during spreading can be expected. This amounts to the same range as the nitrogen losses from mineral fertiliser application. In order to avoid negative impacts of the relatively high salt content of urine, it is recommended to avoid the fertilisation of salt-sensitive plants such as potatoes or fruits, and to rather use it for salt-tolerant plants such as cereals or sugar-beet. If the direct use on edible plants are a concern, urine can alternatively be used as fertiliser for fodder crops or for production of biomass.

One of the challenges of the fertilisation with urine is the variable nutrient concentrations as a result of nutrition, climate, state of health, age and standard of living (Faechem et al., 1983 cited in Fehr, 2007). In order to fully meet the nutrient requirements of the plants, a combination of fertilisation with urine and mineral fertiliser can be useful, since in urine the nitrogen content is comparatively higher than the content of other nutrients (Tidaker, 2007).

As urine is not yet accepted as fertiliser in Europe, it is problematic to use untreated urine. The use of stored, but otherwise untreated urine, does however present a low-cost method which can be applied in economically less developed countries. In order to minimise the (logistic) requirements for collection, transport and application of urine, dilution with flush water should be as low as possible. Several waterless urinals and toilet models that don't use water for flushing urine, are available on the market. Due to the current legislation, urine is used as fertiliser at present only in research projects in Europe. In Germany, there are up to now only scattered implementations of urine-diverting toilets such as the Stranddorf Augustenhof (Bollmann and Bollmann, 2009), the SCST project in Berlin (Peter-Fröhlich et al., 2004) and the GTZ main office in Eschborn (SuSanA (ed.), 2010). Initiated by the NGO SUDEA in the 1990s, Ethiopia was one of the first countries to implement urine-diverting toilets. Nowadays, there are a number of projects promoting the separation and use of urine such as ROSA in Arba Minch (Langergraber et al., 2008) and ESE in Sodo and Awassa (Meinzinger et al., 2009).

Urine can be treated to reduce the volume, facilitate handling and eliminate micropollutants. Another advantage of urine treatment is the generally higher acceptance of the treatment products as fertiliser products when compared to untreated urine. A very effective and well tested way of treating urine is the precipitation of magnesium ammonium phosphate (struvite). Magnesium is added to urine in the form of MgO, $Mg(OH)_2$ or $MgCl_2$. MgO as precipitant is preferable for achieving a high efficiency. Due to the molecular ratio of P:N being 1:1 in the final product, there is still excess of nitrogen in the residual solution after the precipitation process. If this excess of nitrogen is to be recovered, additional processes such as ammonia stripping (see below) or adsorption to zeolite have to be applied. About 95-98% of the phosphorus in the substrate can be bound in MAP (Maurer et al., 2006a). According to Ganrot (2005), potassium and small amounts of sulphur can also be recovered. Potassium is however only bound in MAP if there is no more nitrogen available. Usually there is excess nitrogen in urine or other wastewater-based substrates, so that only very little potassium replaces the nitrogen in the MAP (Udert et al., 2004). The concentration of nutrients in MAP has the advantage of easier transport and storage as well as elimination of pathogens and micropollutants. In addition, there is no ammonia volatilisation during fertilisation and the risk of salinisation of the soil is reduced. According to Römer (2006) plant trials showed that all three components Mg^{2+}, NH_4^+ and PO_4^{3-} have fertilising effects and are value-adding components.

Air or steam stripping can be used to strip out ammonia from substrates such as fresh urine or urine after MAP precipitation. Air or steam is introduced into a stripping column and volatile components such as ammonia are removed from the aqueous phase. The stripped ammonia is subsequently condensed and bound in scrubbers e.g. in sulphuric acid as ammonium sulphate solution. This process has been mainly tested at laboratory scale (see, for example, Behrendt et al., 2002; Tettenborn et al., 2008). The obtained product, an ammonia solution, represents a valuable fertiliser with ammonia concentrations of about 12% in about 1/40 of the initial volume (Tettenborn et al., 2008). The depleted substrate contains only about 2% to 10% of the original nitrogen and can, for example, be discharged together with other wastewater flows for further treatment of organic matter. Other processes such as evaporation, ion exchange or freeze concentration exist, but are not described here in detail. More information is given by e.g. Maurer et al. (2006a) and DWA (2008).

Faeces

The primary consideration when using faeces is their high organic matter content; the nutrient content is rather secondary (see Section 2.4.2). Source separated faeces can represent an important input to agriculture, provided that the pathogen content is reduced. Possible methods for pathogen reduction are, for example, composting (large and small scale) (Niwagaba, 2009), vermicomposting (Gajurel et al., 2007) or chemical treatment with lime or ammonia to increase the pH (Vinneras, 2007). Another option is the use of faeces as input for the production of Terra Preta (Jensen, 2009; Factura et al., 2010). The focus of the assessment in this study is on composting in large scale facilities (see Sections 4.1.8.1 and 5.1.8). Small scale composting often achieves only variable temperature increases, which limits the effectiveness of pathogen die-off. For the composting process faeces should be mixed with organic waste in appropriate ratios to balance the C/N ratio. In addition, a high fraction of easily degradable organics is required to achieve conditions appropriate for sanitisation, i.e. temperatures higher than 50°C (Niwagaba, 2009). Thus, a linkage to organic waste management is provided. Compost is particularly valuable as soil conditioner, but it shows also a fertilising effect. According to Amlinger and Götz (2000), only 5% to 15% of the compost nitrogen is plant-available in the first year with releases of 2% to 8% in the following years. However, with long-term applications the availability of nitrogen for plants is reported to generally increase (Kluge, 2008). The fertilising efficiency of phosphorus and potassium from compost application is considered to be high (Kluge, 2008). Examples of faeces treatment include the Lambertsmühle project in Germany (Oldenburg et al., 2002), and the ROSA project in Ethiopia (Ercolano, 2009).

Blackwater

As for faeces, the main benefit of blackwater is the organic content and the related potential use as an energy source, which is discussed in Section 2.5.2. Blackwater also contains valuable nutrients and can possibly be used as liquid fertiliser or treated for nutrient recovery. Low dilution is desirable, therefore low flush toilets such as vacuum toilets or pour flush toilets are recommended if further use is intended. Anaerobic digestion reduces the pathogen content of blackwater, but thermal or chemical treatment is required for further pathogen inactivation. Thermal treatment can be achieved by heating blackwater up to mesophilic or thermophilic temperatures for an adequate time (Wendland, 2008). As this is a rather energy-intensive treatment process, post-composting, e.g. together with organic waste, is recommended for economically less developed countries; therefore blackwater will then be exposed to a sufficient temperature increase. The anaerobic digestion process results in a mineralisation of nutrients, particularly of nitrogen which therefore become more plant-available (Peretzki, 2006). Easily degradable organic matter is converted to methane and carbon dioxide during digestion, so that the more persistent fraction remains, which is important for the permanent humus content of the soil. According to Peretzki (2006) this also reduces the risk of denitrification and the immobilisation of nitrogen in the soil. Fertilising tests using the effluent of an anaerobic reactor showed very good results in field and greenhouse tests (Simons and Clemens, 2004, cited in Wendland, 2008). In Germany, there are two examples, namely Lübeck-Flintenbreite and Freiburg-Vauban, where blackwater is collected with low dilution in vacuum toilets with the intention to produce fertiliser after anaerobic digestion (Peters, 2002). In Ethiopia, experiences with anaerobic blackwater treatment are only starting, but a possible link to the large-scale SNV program related to agricultural biogas installations exists.

To recover nutrients from blackwater in a more condensed form, treatment processes such as struvite precipitation (see above) or evaporation are feasible. Alp and Otterpohl (2008) report that in laboratory scale experiments evaporation of blackwater achieved a concentration of 80% of N and 95% of P in 6% of the initial volume.

2.5 Energy

Beside water and nutrients, energy is another aspect of resource efficiency related to urban sanitation. The current discussion on sustainable energy management focuses on the one hand on the replacement of fossil energy by renewable energy[30] and on the other hand on increasing the energy efficiency. As the following sections will show, water and sanitation are related to both aspects, since the energy efficiency of current systems can still be improved and wastewater/excreta can also provide a source of renewable energy.

2.5.1 Energy demand for water and sanitation

The relationship between water and energy is manifold. Firstly, hydropower represents a means of power generation. Additionally, water is used for cooling in power generation from fossil fuels. Secondly, energy is required for drinking water provision, i.e. for water treatment and transport. Also wastewater treatment generally requires energy, although low-cost treatment methods as applied in, for example, Ethiopia require only little or no energy.

According to AG Energiebilanzen (2009) the total primary energy consumption per capita in the year 2008 in Germany was 47,268 kWh p^{-1} y^{-1}. This corresponds well with data from the World Bank (2009b) quoting per capita energy use in Germany to be about 4.19 toe p^{-1} y^{-1}, which equals about 48,700 kWh p^{-1} y^{-1}. About a third of this amount is used directly in households, while the balance is energy demand for transport, industry and small businesses (Pauleit, 1998). About 11% of the energy demand of households is for hot water. 75% of the energy demand, and thus by far the

[30] In Germany, about 85% of the primary energy is from fossil sources (UBA, 2009). Besides being limited resources, fossil energy sources contribute to greenhouse effect and global warming. The 2008 amendment to the Renewable Energy Sources Act aims at increasing the ratio of renewable energies up to at least 30% in 2020. In Ethiopia, electricity is generated almost exclusively by hydropower (iea, 2009).

largest fraction, is for heating. The remainder is for cooking, electrical devises and lighting (UBA, 2006).

The water and wastewater systems contribute only little to the overall energy consumption and the related environmental impacts. According to a case study for Berlin, the water supply and wastewater management contributed only 1% of the total CO_2 emissions[31] (Grangler et al., 2001). German wastewater treatment plants require in total about 4.2 $TWh_{el}\ y^{-1}$ (Müller et al., 1999). The approximate electricity demand for wastewater treatment (without sewerage) in Germany is about 40-60 $kWh_{el}\ p^{-1}\ y^{-1}$ plus a heat demand of about 50 $kWh_{th}\ p^{-1}\ y^{-1}$. In wastewater treatment a major portion of the electricity demand is for aeration required for oxidisation of organic matter and nitrogen; this uses about 60% to 80% of the electricity consumption (Müller et al., 1999).

Ethiopia is one of the few countries in Africa with a high potential for hydroelectric power. Currently, electricity production is almost exclusively by hydropower. But the majority of the population is not connected to the electricity grid. According to ENA (2008) national access to electric power in 2008 was about 22%, but plans exist to increase the access to 100% in the next 10 years. Total energy use in Ethiopia is about 0.29 toe $p^{-1}\ y^{-1}$ or 3,373 kWh $p^{-1}\ y^{-1}$ (World Bank, 2009b). Energy consumption for residential purposes is 38%, while that for industry and commercial and public services are 35% and 25% respectively (iea, 2009). Data on energy demand for water and sanitation is not available, but it is presumed that this demand is negligible considering the low use of technical equipment. In this context It is important to note that alternative energy sources such as charcoal, animal manure and firewood were estimated in the 1990s to supply as much as 96% of the country's total energy consumption (LoC, 2009), and these sources still provide the major part of Ethiopia's energy. Even in urban areas per capita use of wood fuel (e.g. firewood and charcoal) is estimated to be on average about 120 kg $p^{-1}\ y^{-1}$ respectively, reaching values as high as 1000 kg $p^{-1}\ y^{-1}$ (Abebaw, 2007). Taking into account a calorific value of 15 MJ kg^{-1} and 30 MJ kg^{-1} for wood and charcoal respectively (EngineeringToolbox, 2005), this represents an energy use of about 1,500 kWh $p^{-1}\ y^{-1}$. Wood fuel use contributes significantly to deforestation, soil erosion and consequently to the loss of productive land. Today, only 10% to 15% of the land is covered by forest because of rapid deforestation in Ethiopia over the last 30 years (LoC, 2009). In addition, traditional stoves impact on health due to the emission of smoke and particulate matter. This emphasises that the generation of energy from alternative sources as discussed in the next section represents an important contribution to the efficient use of resources.

[31] Traffic accounts for about 23% and private households for 33% (UBA, 1998 cited in Grangler et al., 2001).

2.5.2 Energy production

Anaerobic processes allow the use of energy contained in organic matter through the generation of biogas. In principle, substrates such as blackwater (Wendland, 2008), greywater (Elmitwalli and Otterpohl, 2007), mixed wastewater (Seghezzo, 2004) and sewage sludge/faecal sludge (Amuda et al., 2008) can be used as inputs to anaerobic digestion processes. Because faecal matter has a comparatively low energy yield, since it represents already digested material, organic matter can be added to the process to improve performance of the digestion and increase biogas production. The energy value of biogas depends on the CH_4 content, which varies between 55%-75% depending on the digested substrates and other parameters such as temperature, digestion time and pre-treatment. With a calorific value for methane of 35.8 MJ m^3 (Sasse, 1998), the calorific value for biogas is stated to be about 20-25 MJ m^3 (Köttner, 2005; Müller et al., 1999; Thomé-Kozmiensky, 1995). Sasse (1998) furthermore specifies that 1 m^3 of biogas can replace about 5 kg of firewood or 0.6 litre of diesel fuel. Biogas may be used in burners or combustion engines. In Europe, biogas is most often used in combined heat and power plants for the best efficiency. Waste heat can then be used for heating the reactor and also in households for heating and hot water production. Recent developments include the feed-in of biogas into the gas distribution system and the implementation of fuel cells to increase the degree of efficiency. Anaerobic reactors can be also built cost-efficiently, so that application in countries such as Ethiopia is feasible (Kossmann et al., 1999; Sasse, 1998). The first pilot tests and implementations have been done in several locations in Ethiopia showing promising results if sanitation is combined with treatment of organic waste and animal manure (Gairola, 2008; Wagner, 2006). In Germany, energy production from anaerobic digestion of biomass is a wide-spread technology, yet, in only a few cases blackwater is directly used or intended for use (e.g. Peter-Fröhlich et al., 2007; Peters, 2002).

In addition to the biomass content and its chemical energy content, wastewater often also has a thermal value. Heat recovery from wastewater allows the reuse of generated heat to be used for hot water. In principle, heat utilisation is possible from centralised sewer systems where there are recipients close to the sewer, but also on a decentralised level such as households, i.e. directly at the source. The recovered energy can be used for heating but also for cooling purposes[32]. For more information on heat recovery see also Section 4.3.3.

[32] Examples of heat recovery exist in Switzerland for more than 30 years, and this technology is nowadays more and more applied in pilot installations in other countries, too (Schramm, 2008). For Switzerland, it is estimated that about 5% to 10% of all buildings could be heated with heat recovered from

2.6 Economic aspects

Water and sanitation show various connections to economics. On the one hand, inappropriate water supply and sanitation can hamper economic development due to spread of diseases, loss of productive time, pollution of natural resources, etc[33]. On the other hand, the water and sanitation sector provides business opportunities and can contribute to employment generation. Furthermore, water and sanitation require expenditures by private persons and authorities for investment and operation. In this context it should be remembered that "affordability is relative to resources available to a particular community […] and in general relative to GNP of the country" (Grau, 1996, p.96). Grau states that a maximum 1% to 1.5% of the GNP should be spent on sanitation. Other authors, e.g. Pfeiffer (2009), relate sanitation expenditures to GDP and also conclude that if annual costs for sanitation systems are up to 1% of GDP they are perceived as affordable. In 2008 the specific GDP in Germany amounted to about 29,910 € p^{-1}, whereas Ethiopia's GDP per capita was about 4,580 ETB p^{-1} (World Bank, 2009b)[34]. Therefore sanitation systems with annualised costs of less than about 300 € p^{-1} y^{-1} in the case of Germany and 46 ETB p^{-1} y^{-1} in the case of Ethiopia, can be considered to be affordable in the country's respective economic context. Subsidies or cross-subsidies might be necessary to make water and sanitation affordable to all (Pfeiffer, 2009).

The level of treatment strongly impacts on the required costs. For example, nutrient removal is considered to double the cost of simple wastewater treatment plants (Maurer et al., 2006b). Along these lines, simple pit latrines cost only about a fourth on average compared to septic tanks, which in turn cost only about a fifth of conventional sewerage (UNEP and IETC, 2002). It should be noted though that these numbers refer only to direct costs and do not take any indirect costs such as groundwater pollution or health impacts into account.

The German water and sanitation system is characterised by durable infrastructure with high investment costs being depreciated over a long time period. In addition, constant re-investments, maintenance and operating costs, which contribute about 45% of the

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sewerage (Jacquemart, 2005). In Hamburg, the first installation is currently implemented (Werner and Augustin, 2009).

33 Hutton and Haller (2004) conclude in their analysis on costs and benefits of water and sanitation improvements that in developing countries every euro invested leads to 5 to 11fold economic benefits.

34 World Bank (2009b) states 44,000 US$ p^{-1} for Germany and 330 US$ p^{-1} for Ethiopia, which was converted using historical exchange rates (OANDA, 2009).

annual costs, are needed (Maurer et al., 2006b). Operating and capital costs for sanitation are in general required for transport of wastewater e.g. in sewer systems, for rainwater management and for wastewater treatment. Sewer systems are particularly capital intensive as they require about 80% of the investments, but they account also for about two thirds of the total costs of wastewater management (Bode and Grunebaum, 2000; Maurer et al., 2006b). German wastewater management is considered to be relatively costly compared to other European countries[35]. According to Destatis (2009) the average wastewater fee per capita in 2007 amounted to about 150 € p^{-1} y^{-1} and the average water supply fee amounted to about 130 € p^{-1} y^{-1}.

Ethiopia's water and sanitation sector is severely hampered by lack of funds. An estimated 300 million US$ are required per year to meet the MDG on water and sanitation, of which only one third is expected to be available from government (12%), communities (15%) and donor allocations (73%) (AMCoW et al., 2007). Financing mechanisms such as fee collection are still rather fragmentary, particularly for sanitation, and the dependence on external financing is high. In future (private) service providers are increasingly expected to raise capital through the collection of fees, and to increase the ratio of cost recovery. Average data on household expenditures for water and sanitation is not available. An indication of costs for water supply and sanitation is nevertheless given in the section on Arba Minch (Section 3.6.1).

[35] Bode and Grunebaum (2000) argue that the comparatively high wastewater fees in Germany are a result of high degree of cost recovery, high standards as well as high labour and energy costs.

3 Method and case studies

This chapter illustrates the method of the study and introduces the selected case studies. Firstly, available methods and tools for assessing the resource efficiency of wastewater systems are briefly discussed. Thereafter, an overview of the particular method used in this study, is given. The selected method is based on a Material Flow Analysis (MFA) that includes mass, water and nutrient flows, enhanced by energy and cost analyses (in this study called ceMFA). The chapter furthermore illustrates general aspects of the applied modelling procedure and data collection. Subsequently, in Sections 3.5 and 3.6 the case studies Hamburg and Arba Minch are introduced and the system boundaries selected for this study are presented. Finally, the investigated systems are described for the respective case studies.

3.1 Overview of systems analysis methods

This section provides a theoretical background for the selection of a Material Flow Analysis as assessment tool, by briefly discussing different environmental and economic assessment methods.

3.1.1 Material Flow Analysis (MFA)

One of the most general and elementary ways of analysing a system and its environmental impacts is by means of a Material Flow Analysis (MFA). MFA is a tool that allows the study of the flow of materials (i.e. substances or resources) being used and transformed within a system defined in space and time. The sources, pathways, as well as the intermediate and final sinks of a material are included in this analysis (Brunner and Rechberger, 2004). The systems under consideration can consist of one or several particular processes. Processes are usually defined as black-box models, and include the transformation, production and consumption of materials. Material Flow Analysis is based on the law of mass conservation, comparing all inputs, stocks and outputs of a process and using mathematical models to describe the processes and flows.

Material flow analysis involves the application of a systems approach for the purpose of planning and decision making. For example, on a spatial scale such as regions or countries, it can be used to improve strategic planning and public policy. Weak points can be identified and possible scenarios can be assessed. MFA can be used for environmental and economic accounting, monitoring, as well as planning and evaluating policies. It can help in recognising resource demand and environmental impacts in advance, and evaluating the effects of different technical or strategic measures. The scope of MFA studies is wide and ranges from global systems, e.g. global phosphorus metabolism (Liu et al., 2008), to country level, e.g. resource consumption in Germany (Bringezu et al., 2008), to household level, e.g. waste flows in households (Binder and Mosler, 2007). MFA has also been applied in waste, water and wastewater management[36].

Different software tools are available for material flow analyses, although for simple MFA with only few processes, spreadsheet software such as Microsoft Excel can be used. Besides some Life Cycle Assessment (LCA) software packages (see Section 3.1.2), which can be used for MFA as well, there are some software tools specifically designed for MFA. Examples include Software for Substance Flow Analysis (STAN) developed by Vienna University of Technology, or the Swedish MFA application in waste management called ORWARE and its adaptation for water management URWARE (Jeppson and Hellström, 2002). In this study, the software SIMBOX developed by EAWAG (Switzerland) is used. This package allows a high degree of adaptation to the user's needs (Bader and Baccini, 1996).

3.1.2 Life Cycle Assessment (LCA)

Often an MFA is considered to be the basis for Life Cycle Assessments (LCAs). An LCA follows a structured methodology for analysing the environmental impacts of a good or a service along its whole life cycle. A life cycle generally includes raw material extraction (i.e. preceding processes), manufacture, distribution, use and final disposal. This approach is usually referred to as a cradle-to-grave approach. The use of functional units such as the treatment of a certain amount of water, enables comparisons between different alternatives. The procedure for LCA is internationally standardised[37]. There

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[36] Examples of MFA applied in water and wastewater management include, among others, case studies in China (Huang et al., 2007), Denmark (Magid et al., 2007), Germany (Grangler et al., 2002), Ghana (Forster et al., 2003), Vietnam (Montangero et al., 2004) and Zimbabwe (Gumbo, 1999).

[37] The four main phases of an LCA are defined according to ISO standards 14040 and 14044 as follows:

- Goal and scope definition (including description of functional unit, system boundaries and method applied) are different LCA software packages available[38]. One advantage of these packages is their ease of use. There is however a dependency on the modules integrated into the various packages and usually the options for adapting the software to specific needs are limited.

LCA is particularly useful for supporting companies' business strategies, as well as research and development (Cooper and Fava, 2006). It is now also increasingly used for system analysis in strategic planning of service provision. A list of early LCA studies in water and wastewater management is given by Lundin (2003). Most of these studies focused on the level of processing at treatment plants, but did not include a system perspective (e.g. Dennison et al., 1998; Pettersson, 2001; Pons et al., 2004). Examples of more system-oriented LCAs include the work of Lundin et al. (2000) and Remy and Ruhland (2006). In general the studies concluded that energy, as well as the emissions of nutrients and heavy metals, are important factors with regard to environmental impacts.

Drawbacks of the LCA method include the need for large amounts of data and the aggregation of results into impact categories, which implies a certain loss of information. Furthermore, the assessment is confined by the system boundaries and for example, changes in the system, such as changes in demand, are not easily accounted for. This means that dynamic development processes such as urban development (i.e. changes in land use or demographic changes) are difficult to incorporate since only standardised conditions are used. In general the focus of LCA is rather on specific goods or services, whereas the application of MFA often focuses on countries, regions or sectors.

3.1.3 Other environmental assessment methods

An adaptation of the LCA approach is the Life Cycle Energy Analysis (LCEA), which includes all energy inputs to a product or service, including any manufacturing processes. If only the operational phase is assessed in an LCA the analysis resembles an Environmental Impact Assessment (EIA). An EIA can be defined as "a process of

- Life cycle inventory (including data collection, description and verification of data, modelling of the system usually based on mass and energy balances)
- Life cycle impact assessment (definition of environmental impact categories, can be normalised and weighted)
- Interpretation (including analysis of major contributions, sensitivity and uncertainty analysis, conclusions)

[38] In the international context GaBi is the software most often used (Cooper and Fava, 2006), whereas in Germany UMBERTO seems to be the most widely used software.

identifying, predicting, evaluating, and mitigating the biophysical, social, and other relevant effects of proposed projects or plans and physical activities prior to major decisions and commitments being made" (Kirk et al., 2005, p.5-3). In Europe, EIA comprises standardised procedures to assess the environmental effects of projects on aspects of the environment such as fauna, flora, soil, water and air, but also human beings and the landscape (EEC, 1985).

Another approach to analyse material and energy requirements is Material Intensity per Unit Service (MIPS). The focus is also on one or a small number of specific products or a service delivery, for which cradle-to-grave or life cycle impacts are identified. The result is a material intensity[39] needed to provide a specified level of service. The MIPS concept was applied in wastewater management for example by Reckerzügl and Bringezu (1998). The MIPS approach is particularly useful to identify environmental impacts, but does not for example, show any potential for recycling.

An overview of further techniques and approaches for assessing eco-efficiency, such as Total Material Requirement and Output (TMRO) or Ecological Footprint Analysis, is given by Daniels (2002).

3.1.4 Economic assessment methods

There are a variety of methods available by which an economic evaluation can be done. Examples of such valuations are cost comparisons, cost-benefit analyses or life cycle costing, which can be applied for system comparisons. Most often simple cost comparisons based on e.g. net present values are applied for comparing sanitation systems[40]. Systems must achieve the same benefits if a simple cost comparison is used. If this is not the case, a cost-benefit analysis can be applied, which includes monetary values of the expected benefits of a system. Similar to the cost-benefit analysis is the so-called cost-effectiveness analysis, where costs are compared to the outcomes or effects of a system under consideration; these costs are not measured in monetary values. This has the advantage that benefits, which are often related to different dimensions (i.e. environment or society), do not need to be valued in terms of money. However, targets need to be clearly specified in order to evaluate the relative achievement of objectives by the systems under comparison. A combination of cost-effectiveness analysis and

[39] The material intensity is classified into five categories: abiotic materials, biotic materials, soil, water and air.

[40] Examples of cost comparisons of wastewater systems that include nutrient recovery, are the studies done by Oldenburg and Dlabacs (2007), von Münch and Mayumbelo (2007), Wiecenec (2004) and Prager (2002).

cost-benefit analysis together with a material flow analysis is presented for a waste management assessment by Döberl et al. (2002).

In principle, all kinds of costs and benefits could be included in a cost analysis. Social and environmental costs or benefits (i.e. intangibles) are however, usually difficult to quantify and are therefore often not included in economic evaluations[41]. However, this must not lead to the selection of cost-effective alternatives that are a burden to society or the environment. In addition to purely financial criteria, there are also spillover effects related to economic effects. These include for example, employment generation, poverty alleviation, reduced use of foreign exchange or imported goods, and also improved environmental management. Often, these effects are difficult to quantify, but it is important for any decision making that these spillover effects are taken into consideration. In addition, cost-benefit comparisons involve an underlying conflict in terms of scale, since the cost analysis is preferably done on a relatively small scale for improved accuracy. On the other hand, benefits need to be looked at from a wider point of view spatially and temporally, because effects such as downstream water quality can play a decisive role (Prager, 2002).

3.1.5 Multi-criteria assessment methods

Where several criteria from different categories need to be integrated, multi-criteria analysis may be applied. Different methods exist to support multi-criteria decision problems. For example, Hiessl et al. (2003) and Malisie (2008) applied the Analytical Hierarchy Process (AHP) for their sustainability assessments of different wastewater management options. In AHP assessments ranking is done by pairwise comparison of the criteria to obtain a weighting and to allow subsequent comparison of the alternatives. The PROMETHEE approach (Preference Ranking Organization METHod for Enrichment Evaluations), an outranking method, has also been applied in decision problems in water management (Heinrich, 2001; Peters et al., 2003). Another example of multi-dimensional decision support is the so-called utility analysis. Here, an objective system with measurable criteria is set up, against which the systems under consideration are scored. Usually, the criteria are weighted to reflect the differences in relevance and are usually scored against the monetary costs. Another approach for solving multi-criteria problems is the aggregation of criteria to one common unit. For example, they can be expressed in energy or cost equivalents. An exemplary study of

[41] Different methods exist to calculate monetary values for intangibles, including direct methods such as contingent valuation and indirect methods such as travel cost method or avoidance method (see Hanley and Spash, 1993 cited in Starkl et al., 2004a). Yet, often relatively large uncertainties are implicit in these kind of evaluation methods.

this in the field of wastewater management, is the work done by Dockhorn (2007). He translated benefits such as recoverable nutrients into their economic potential and could therefore aggregate costs and benefits into monetary equivalents. A thorough analysis of available assessment methods in wastewater management, including multi-criteria decision support, is given by Starkl et al. (2004b). They conclude that, depending on the scale and the type of the decision problem, different methods might be useful. In general, they recommend the use of simpler methods because of the increased transparency and traceability.

Wherever a weighting of criteria is required, this should be done by decision makers and other stakeholders, in order to reflect the priorities of those concerned. Due to the weighting procedure the modelling results undergo a subjective ranking. In addition, the result of the ranking procedure is usually a single value, combining the different criteria. This single value often conceals the very complex underlying assumptions and input parameters. Therefore, in order to keep the results more transparent, aggregation and ranking of criteria are not used in this thesis.

For all methods it needs to be mentioned that a thorough assessment is dependent on the availability of reliable data. This can be a problem in developing countries where data collection is sometimes afflicted by high uncertainties.

3.2 Research method – Cost, Energy and Material Flow Analysis (ceMFA) as assessment tool

The objective of this thesis is the comparative assessment of different urban wastewater management options[42], which allow a recovery of nutrients. Material Flow Analysis is selected as principle research method due to its comprehensive approach and is combined with energy balances and cost estimates. A Mathematical Material Flow Analysis (MMFA) (sometimes referred to as Material Flow Analysis modelling) is used. An MMFA includes the extension of a classical MFA with modelling concepts to allow modelling with more uncertain data (Baccini and Bader, 1996). Thus, data from literature values, complemented by plausible estimations can be used instead of data from extensive measurement campaigns (Schaffner, 2007). Material and substance flows

[42] The alternatives are referred to as options or systems and not as scenarios, since the term scenario usually relates to different projected future events in terms of socio-economic or environmental developments. The focus here is purely on technical variations for the achievement of nutrient recovery.

are assessed using a quasi-stationary[43] model, i.e. changes in the flows over time are not reflected. The temporal boundary refers to one year. Preceding processes such as raw material extraction or manufacture are not included.

This study includes the following main steps for the mathematical material flow analysis (see also Baccini and Brunner, 1991; Baccini and Bader, 1996):

1) System analysis based on (a) specific objective(s), including the definition of system boundaries, substances to be considered, processes and flows
2) Set-up of an equation system to describe the system behaviour
3) Parameter definition for the determination of variables
4) Data collection
5) Simulation runs including calibrations, uncertainty and sensitivity analyses
6) Analysis, interpretation and presentation of the results

The approach used in this study is the integration of not only mass and nutrient flows in the MMFA, but also of energy and economic aspects. This allows the modelling of energy flows and cost estimates in direct relationship to varying substance and material flows. This method is called a cost, energy and Material Flow Analysis (ceMFA) in the context of this study. The flows are generally calculated as total flows and later converted to specific flows, such as flow per capita.

3.2.1 Mass and nutrient flows

The main component of a ceMFA is the modelling of mass and nutrient flows. Mass flows include not only goods such as organic waste or chemicals used in treatment processes, but also water flows. Therefore, the mass balances can also be used for assessing water balances and water-related issues such as groundwater extraction.

The nutrients assessed in the ceMFA include nitrogen and phosphorus for both case studies, as well as potassium and sulphur for Hamburg only. These nutrients are selected due to their relevance for water protection and agricultural production (see also Section 2.4). The reason that potassium and sulphur is not modelled for the case of Arba Minch is the lack of data on these two nutrients in the Ethiopian context. In addition, potassium and sulphur are not (yet) applied as mineral fertiliser in Arba Minch and are, therefore, not yet given high importance by relevant stakeholders and decision-makers in Ethiopia. Also environmental impacts such as eutrophication are not

[43] The quasi-stationary model implies that the flows and any stock rate changes are constant over time. This means that changes in a balance volume are linear (Baccini and Bader, 1996).

related to those two nutrients. The nutrients N, P, K and S are analysed as basic elements (i.e. as total nutrients in organic and inorganic forms) even though they exist in a variety of compounds in urban water systems. Compounds however, usually show rather dynamic properties; this is not reflected in the selected modelling approach. Dynamic modelling would require detailed knowledge of the internal structure of the modelled processes (Dalemo et al., 1997), which is beyond the scope of this study due to the focus on overall performances of the systems.

In addition, total organic carbon is assessed in the ceMFA. This is, on the one hand, due to the importance of organic matter as a pollutant, i.e. an eutrophying substance in surface waters. On the other hand, organic carbon shows a direct relation to the energy balances. Furthermore, organic matter is considered a valuable soil conditioner in agriculture.

3.2.2 Energy analysis

The energy analysis, which is based on a stationary model, is directly coupled with processes and flows of the ceMFA. This means that mass, nutrient and organic carbon flows of the different options are used as input values to arrive at energy balances. Energy demand as well as energy production (e.g. through production of biogas or incineration) is assessed. Only those processes that show differences in the investigated options, are included. Therefore energy values are not absolute values, but represent relative values. For example, the overall energy consumption of households is not included; only where processes related to the urban water system (e.g. decentralised anaerobic digestion to replace firewood) impact on the energy demand, are these aspects considered in the modelling. Grey energy or embodied energy[44], which is the energy used for an entire product life cycle, is not included within the scope of this study. This means that only the energy needs for operation are accounted for; energy requirements for production of hardware are excluded.

Energy balances are given as primary energy. Therefore, other energy carriers such as electricity or methane are converted to primary energy to reflect the efficiencies of related processes. For the conversion of electricity consumption into primary energy demand, a factor of 2.7 is used for the case of Germany (DIN, 2007a) and 1.5 for the case of Ethiopia (based on Adam, 2009)[45]. The difference derives from the difference in energy mix used for each case study; the Ethiopian energy mix depends almost

[44] Some authors refer also to the term "cumulative energy demand" (CED).

[45] Adam (2009) states an efficiency for hydropower of 70% and for power transmission of 95%.

exclusively on hydropower, which has a higher efficiency than the German energy mix. For fuel and natural gas a primary energy factor of 1.1 is used (DIN, 2007b). Further energy-specific parameters are listed together with the respective process descriptions, as well as in Annexes A and F.

The proper SI unit for energy analyses is Joule. However, in this study the unit kWh is used because of its wider prevalence in daily routines.

3.2.3 Economic evaluation

The main aim of the economic evaluation in this thesis is to give an order of magnitude of total costs and a rough ranking of different cost pools. Although absolute values are stated, it needs to be kept in mind that cost ranges in this study represent indicative costs rather than definite costs. An economic perspective and not a financial perspective (e.g. from a utility's point of view), is taken. Overall benefits for the economy such as health impact, food security or employment generation are not included in the analysis. This is in line with the recommendations of KfW (2008) for a simplified economic assessment of better sanitation service.

The cost analysis is carried out as a dynamic cost comparison using annuities. German guidelines for dynamic cost comparison and for economic appraisal of building installations are used as basis for the calculations (LAWA, 2005; VDI, 2000). The annuity of each system is the sum of annualised present values of investment and reinvestment costs and operation costs. The conversion of investment costs into annuities is based on differentiated life spans for the various components. In general, life span for civil works is estimated to be between 40 and 50 years, while for mechanical parts a life span between 10 and 12.5 years is assumed. For discounting a real interest rate of 3% is used for the case of Hamburg (as recommended by LAWA, 2005) and 1% for the case of Arba Minch (World Bank, 2008)[46]. To assess the effect of different values, the interest rate is varied in the sensitivity analysis. Operational costs include labour, energy (electricity and heat), material, maintenance and repairs. Monetary benefits are not directly included in the cost comparison, but the respective discussions of the systems comprise an appraisal of benefits.

Fixed and variable costs are assigned to the physical flows and processes as used in the MFA. This allows the identification and allocation to cost centres (e.g. wastewater

[46] Ethiopia's real interest rate is currently negative since inflation exceeds nominal interest rates. This trend is considered to be a risk for healthy economic development (IMF, 2008). Therefore, in this study the real interest rate of 2005 is used.

management or households). Unit costs and specific life spans are listed in the relevant sections of chapters 4 and 5 and Annexes A and F. Costs are adjusted to exclude any taxes or subsidies. Average conditions are considered, and cost ranges are included in the analysis to allow for uncertainties. Wherever possible, cost functions for process costs, which are developed on the basis of power laws or polynomial functions relating costs to process size (e.g. volume, population equivalent, area, etc.), are used. Cost items that are constant in all systems, such as in-house drinking water pipes, are not included in the cost calculation. Therefore, the results show relative advantages or disadvantages, and not absolute values.

Costs differ, depending on whether a system is integrated into a pre-existing infrastructure or into new developments. The cost model for the Hamburg case study includes two setups, i.e. existing infrastructure and greenfield development. For the case of Arba Minch no differentiation is made in this regard, since currently no extensive infrastructure is in place.

The conversion of present values of investments or capital costs to annualised costs is done using the following formula (see also LAWA, 2005):

$$AC = C \cdot \frac{r \cdot (1+r)^n}{(1+r)^n - 1} \qquad [€\ y^{-1}] \text{ or } [ETB\ y^{-1}] \qquad (3\text{-}1)$$

with

AC: annualised capital cost [€ y^{-1}] or [ETB y^{-1}]

C: total capital cost [€] or [ETB]

r: (real) interest rate [-]

n: life span of investment [y]

Varying cost ranges are one of the challenges of economic assessments. Reicherter (2001) states that in wastewater treatment the specific costs for the same type and size of treatment plant can differ up to 150-200% from the average. Therefore, it is useful to indicate cost ranges and to consider any data on costs as indications rather than fixed values. The German standard DIN 267 "Costs in Civil Engineering" differentiates between four phases of costing over the time span of the project planning. According to Reicherter (2001) the level of detailing increases between the phases and results in the following precision ranges:

1. Cost estimate: ± 30%
2. Cost calculation: ± 15%
3. Costs after award of contract: ± 5-10%
4. Final costs: ± 0%

Another differentiation of cost estimation is done by Clark and Dorsey (1982, cited by Chugg, 2007) using following types:

1. Order of magnitude estimate: useful for approximate or so-called ballpark figures – often used to double check other methods
2. Study estimate: unit process estimates are commonly used
3. Preliminary estimates: individual and independent estimates of each item involving a higher degree of accuracy
4. Definitive estimate: requires detailed design estimation and is costly and time-consuming to prepare
5. Detailed estimate: site specific estimate including complete design data – used to control cost during construction.

As this study represents a first appraisal of different systems, the economic assessment carried out within the framework of this study corresponds to a cost estimate or study estimate. Therefore, unit process estimates are used and data uncertainty of up to ±30% seems acceptable. Due to the associated data uncertainty the economic assessment should not be used for ruling out certain options, but should be seen rather as an indication of where system adaptations or more accurate cost calculations are required.

3.3 Modelling approach

For the ceMFA mass, nutrient and energy flows as well as cost estimates are transferred into mathematical models through the formulation of system equations. For the two case studies Hamburg and Arba Minch two different approaches for setting up the system models are followed. For Hamburg, every system is represented in one particular equation system. This is done since the systems differ relatively widely. For Arba Minch, there is only one equation system set up for all systems, which are then specified by system parameters that are introduced to reflect the system characteristics. The latter approach has the advantage that different systems and transition options can be directly compared, using only one equation system for the modelling.

In general, the selection of the system boundaries is crucial for a system analysis. System boundaries refer to spatial and temporal boundaries as well as to the processes that are taken into account (Sonesson et al., 1997). The selection of the system boundaries impacts on the results and their interpretations, as well as on the data requirements (Dalemo et al., 1997). Thus, they need to be carefully defined on the one hand so as to not to limit the system analysis and to allow a wide view on the respective topics. But on the other hand, the scope of the analysis still needs to be manageable in terms of data collection and the amount of required work.

The systems are made up of relevant processes and flows. Processes can be defined as any activities that transform, transport or store materials. Flows are defined as the links between the processes. The processes that are included in the system boundaries must reflect the whole chain of actions for the provision of a certain service, i.e. in this study the urban water and wastewater system with integrated nutrient recovery. The following general processes are therefore included in the system boundaries[47]:

- water supply
- rainwater management
- domestic sanitation, i.e. wastewater management including the management of residues such as sludge
- management of organic solid waste
- agriculture (water and nutrient management)

As spatial system boundaries for both cases, Hamburg and Arba Minch, the administrative town borders are taken into account. However, referring to the Process Agriculture the system boundaries are extended, since the agricultural areas within the towns' borders are relatively small and not sufficient to feed the whole population. Therefore, an area large enough to feed the population of the respective city is considered, i.e. a concept of hinterland is introduced. This is done in order to minimise exports and imports across the system boundaries, and to highlight the urban population's food and fertiliser requirements. The hinterland area is calculated according to the specific food demand and corresponding area required for crop production; this area differs for the cases of Germany and Ethiopia. This calculation is explained in detail in Sections 4.1.1 and 5.1.1.

The processes and flows, i.e. the variables identified in the system definitions, are linked in a set of equations. Initially, balance equations of the different processes are set up, defining the input flows, outputs flows and stock rate changes of every process. In general, this follows the law of mass conservation by adhering to the following relationship:

$$\textit{inflows} + \textit{production} = \textit{outflows} + \textit{accumulation} \tag{3-2}$$

Production and accumulation can be combined as stock rate change, which results in the following general mathematical expression for the balance equations (Equation 3-3). A more detailed description of the derivation of system equations can be found in the

[47] Industry is not included in order to disregard peculiarities and to increase comparability and generalisability. In addition, it is argued that industrial water and wastewater management can be considered in a relatively detached manner from domestic water, wastewater and nutrient management.

report by Baccini and Bader (1996). The algorithm, which SIMBOX uses for solving the equation system, is also described in detail in the same report.

$$\dot{M}_j^n = \sum_i A_{i,j}^n - \sum_k A_{j,k}^n \qquad (3\text{-}3)$$

with

$\dot{M}$: stock rate change (i.e. accumulation or degradation)[48]
A: flow
n: substances, i.e. mass, water, N, P, K, S, C
i: input
j: balance process
k: output

Next, model equations are defined to represent the behaviour of the system in a mathematical way. Processes either modify the material under consideration (i.e. include stock rate changes such as accumulation or depletion), or they are transfer processes. Parameters are used to describe the key characteristics of the systems. For example, transfer coefficients describe the partitioning of a substance in a process, i.e. the transfer of inputs to outputs. This is defined for each output of a process and is not only substance-specific but also technology-specific. Model equations to calculate flows based on transfer coefficients, are generally set up according to Equation 3-4.

$$A_{j,k}^n = tc_{j,k}^n \cdot \sum_i A_{i,j}^n \qquad (3\text{-}4)$$

with

tc: transfer coefficient, with $\sum_k tc_{j,k}^n = 1$

In addition, a substance flow (e.g. nutrient flow) induced by the flow of a good (e.g. wastewater) can be calculated by multiplying substance concentrations with the mass flow of the good as per Equation 3-5.

$$A_{j,k}^n = m_{j,k} \cdot c_{j,k}^n \qquad (3\text{-}5)$$

with:
m: mass flow of good
c: substance concentration [in %]

[48] In the quasi-stationary case, any flows and stock rate changes are assumed to be constant.

If substance flows are directly available (e.g. nitrogen load in excreta per person and day), these are also used directly or they are appropriately converted[49] into total flows. Modelling of energy and costs is interlinked with the modelled physical flows, i.e. mass or nutrient flows. This means that energy and cost variables are introduced and equations are defined by which energy and cost flows can either be calculated directly (i.e. in the case of fixed costs or energy demand) or as dependant on their relationship with mass and nutrient variables (i.e. variable costs and energy demand).

The modelling is complemented by an uncertainty analysis based on Gauss' law of error propagation using Taylor series. For this, parameters are assumed to show normal distribution (Gaussian distribution) and small uncertainties. Gaussian error propagation is then used to determine the error or uncertainty of the variables produced by the interacting parameters. The results of the uncertainty analysis are indicated as error margins of the variables. In addition, SIMBOX allows Monte Carlo simulations to determine uncertainties resulting from parameters with probability distributions other than a Gauss distribution. Therefore, the distribution of every parameter is assigned as being normal, lognormal or uniform; this is done by applying best knowledge for each parameter[50]. Standard deviations (or minimum/maximum values for a uniform distribution) are selected. Monte Carlo samples including random numbers are generated and applied to the parameters, followed by simulation runs. This allows a more precise calculation of the probability distribution of the variables, provided that good knowledge about the distribution of the parameters is available. For the purpose of this analysis Monte Carlo simulations are carried out for every equation system and the resulting uncertainties are compared with the uncertainties based on normal distributions.

Sensitivity analyses are carried out to determine the most sensitive parameters. That means parameters are identified that contribute to a high degree of uncertainty in important variables. The data quality of these parameters can then be refined in order to decrease the uncertainty of the variables. In addition, the range of the change in a variable when the parameters affecting it are changed, is evaluated. The sensitivity analysis calculates the first order changes in variables due to changes in parameters. "Relative sensitivities per 10%" (see Equation (3-6) which is based on work of Baccini and Bader, 1996) can be considered high if the relative change in the variable is higher than the assumed relative change in the parameter. Therefore, the sensitivity analysis is

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[49] Conversions include for example, multiplication by the number of persons or the number of days in a year.

[50] Truncated normal and truncated lognormal distributions can also be assumed.

used to identify key parameters and to get a better understanding of the system as a whole.

$$\text{Relative sensitivity per 10\%} = \frac{\Delta X_i(\Delta p_j)}{\Delta p_j} \cdot \frac{p_j \cdot 0.1}{X_i} \tag{3-6}$$

with:
X_i: variable
p_j: parameter

Furthermore, parameter variations are included in the analysis to show the effect of parameters varying over a larger range. This allows the evaluation of measures with regard to their impact on specific flows or other variables such as specific costs or energy demand.

3.4 Data collection

In general, data for an MFA can be obtained from literature, by assessments based on assumptions or cross-comparisons, or by direct measurements. The quality of the results depends strongly on the data quality used for parameter definition. Since the investigated systems of this study have up to now not been realised in full-scale projects, data from existing systems is not available. At this stage field data is only available from pilot projects. Sources for data used in this study are primarily literature reviews, and to some extent key informant interviews. For the case of Arba Minch very little reliable secondary data is available; this is a barrier to many MFA applications often encountered in developing countries (Montangero and Belevi, 2007). Thus for the case Arba Minch, field surveys are additionally drawn upon; these include questionnaires, focus-group discussions and expert interviews carried out within the framework of the ROSA project. In addition, missing data is calculated by means of mass balances.

The base year for this study is set at 2007, although not all data was gathered from this year. Therefore, conversion factors such as price indices are applied where this was considered necessary. In order to reflect data ranges or uncertainties, approximated standard deviations are integrated into the database. These are derived either directly from literature or from estimations. Error distributions are approximated for Monte Carlo simulations. The uncertainty analyses, as explained above, can then be used to assess the quality of the results based on input data quality.

Due to the scarcity of information from full-scale installations, cost assessments in this study are particularly affected by a rather uncertain database. Therefore cost data is collected from other studies or from pilot projects and checked for applicability.

However, unit costs for innovations usually show an experience curve (sometimes also called learning curve). This means that with increasing units of production costs decrease significantly due to better use of equipment, labour efficiency, standardisation, etc. In addition the effect of economies of scale applies. One of the recent examples of experience curves in wastewater treatment is the cost development of membranes[51]. However, the ex-ante quantification of such developments is difficult. Although it would therefore be worthwhile to consider learning curves for innovations such as vacuum toilets or urine-separation toilets in more detail, this effect is not included in the analysis. However, approximations that take into consideration the general decrease in the cost of innovations, are used in the analysis.

3.5 Case study Hamburg

In the following sections the case study Hamburg is introduced. After a general background of the city and its situation with regard to water supply and wastewater management, the system boundaries of this study are introduced. Subsequently, the systems under consideration are briefly explained.

3.5.1 Introduction to Hamburg

The Free and Hanseatic City of Hamburg is located in the North of Germany (about 53.5° Northern latitude and 10.0° Eastern longitude) (see Figure 3.1). With about 1.7 million inhabitants[52] it is the second-largest city in Germany. Hamburg is one of the few areas in Germany where a growth of the population (particularly from migration) is expected. It is anticipated that by 2020 the population will increase to more than 1.8 million people (Statistisches Amt für Hamburg und Schleswig-Holstein, 2008c). There are about 960,000 households in Hamburg, resulting in a relatively low number of 1.8 persons per household. Since the total administrative area amounts to about 755 km^2, the arithmetical population density is 23 p ha^{-1}. Yet, the actual population density varies greatly, reaching values as high as 180 p ha^{-1} (Statistikamt Nord, 2010).

Hamburg's climate is moderate and influenced by the proximity to the sea. Average air temperatures are between 2°C in January and 18°C in July and the mean annual rainfall

[51] It should be noted that experience curves do not consider the effect of political and regulatory frameworks (e.g. stricter emission standards), which can have a great effect on costs and market penetration (Öko-Institut and Partner, 2004).

[52] For the assessment the number of registered inhabitants in 2007 is used, i.e. 1,741,182 persons (Statistisches Amt für Hamburg und Schleswig-Holstein, 2008b).

is 774 mm (Statistisches Amt für Hamburg und Schleswig-Holstein, 2006). About 8% of the total area is water and Hamburg is characterised by several river courses within its town borders. The largest river is the Elbe River, which has its source in Czech Republic and discharges into the North Sea about 100 km downstream of Hamburg. The port, which is the second largest port in Europe, is considered to be one of the most important economic drivers in Hamburg. Water is an important element for the town for recreational and particularly industrial purposes. The quality of the surface water is therefore a major concern and efforts have been made on various levels to improve the water quality[53].

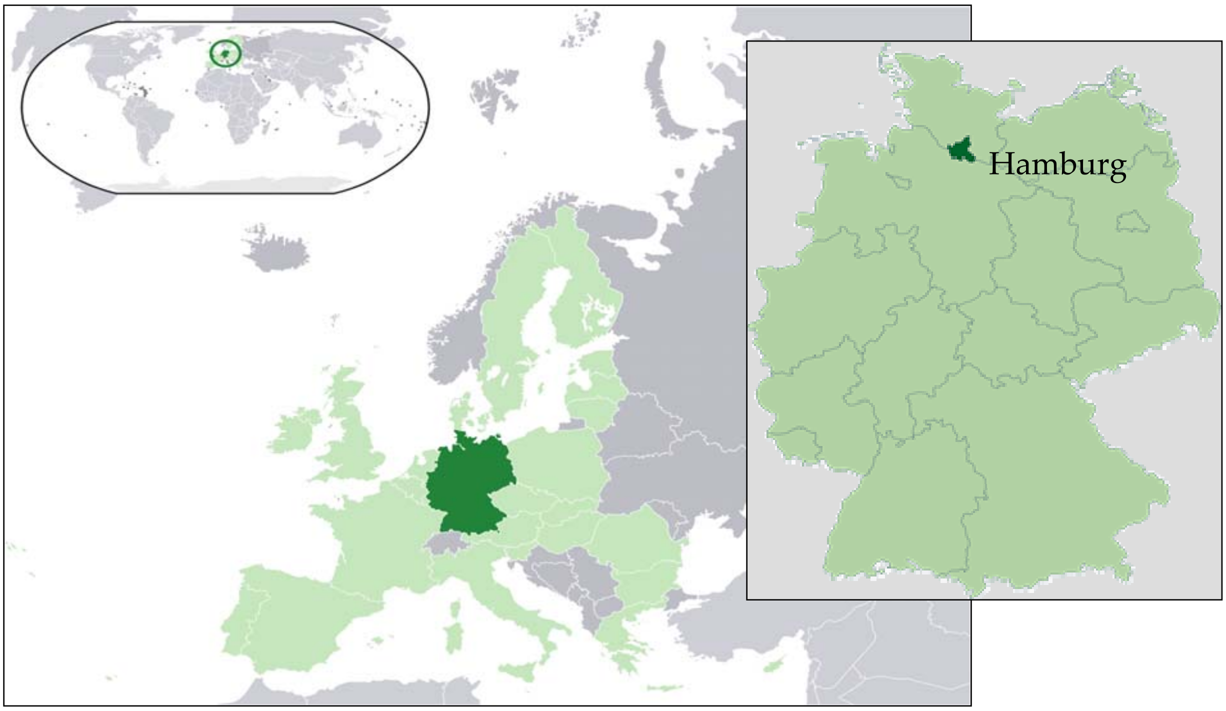

Figure 3.1: Location of Hamburg
(Based on maps from Quizimodo and David Liuzzo, Wikimedia Commons)

Agriculture is practiced on about 25% of the total area of Hamburg, particularly for fruit-growing and flower farming. Yet, the economic importance of the farming sector is rather marginal. The surrounding area however, is shaped by agricultural activities, thanks to fertile soils. In the Hamburg Metropolitan Region, which covers about 20,000 km^2 including the city itself, agriculture is the greatest user of land area (Drücker et al., 2006).

[53] Examples of relevant initiatives are the River Basin Community Elbe, which is concerned with the European Water Framework Directive, and the combined sewer overflow programme of Hamburg Wasser implemented in the last 20 years.

In the past, drinking water supply for the town was sourced mainly from surface water, but since 1964 only groundwater is used. To fulfil the water demand, groundwater is also extracted from areas outside the town borders; these areas include ecologically sensitive areas. The increasing demand is the reason for disputes between Hamburg and the surrounding communities (Stemmler, 2009). Eighteen treatment plants for groundwater treatment and distribution exist. With a total length of more than 5,500 km, the water distribution network is one of the largest in Germany distributing about 350,000 m^3 water every day (Leonhardt, 2005). Water consumption in households has declined due to the introduction of water meters and water-saving installations, to about 107 $l\ p^{-1}\ d^{-1}$ (Hamburg Wasser, 2007) and is expected to further decrease in the future (Maass, 2008).

Hamburg was the first city on the European continent where a sewer system was implemented. The sanitation system in Hamburg in the 19th century was similar to that of other European cities, namely a system of pit latrines, soakways and open disposal. After a disastrous fire in 1842 a complete renewal of the inner city was initiated, including the construction of a sewerage system. Starting with 12 km of sewers at that time, the sewer system in Hamburg now totals about 5,400 km.

Treatment of wastewater though, was neglected for a long time, until a Cholera epidemic in 1892 forced the town's decision makers to start thinking about treating the wastewater formerly discharged as raw sewage into the water courses. Initially several distributed treatment plants were installed. In the 1960s two combined treatment plants in Köhlbrandhöft/Dradenau gradually made the decentralised plants redundant (Eich and Wierecky, 2002). Nowadays, 99.8% of the inhabitants are connected to these plants. The remaining inhabitants rely on on-site treatment of their wastewater (BSU, 2009). In 77.5% of the sewered area totalling 407 km^2, separate sewers have been installed, while 22.5% of the sewered area discharges into combined sewers. The treatment plant currently treats about 168 million m^3 of wastewater every year, using physical, biological and chemical treatment processes. About 35% of this volume is from rainwater and 8% is from surrounding communities (Hamburg Wasser, 2007). The effluent is discharged to the Elbe River. Sewage sludge is anaerobically digested, dried and incinerated in a mono-incineration plant installed in 1997.

Organic waste (kitchen and garden waste) is currently collected separately only in some parts of the city (about 22% of the households), since home composting is recommended in areas with plot sizes larger than 600m^2 (BSU, 2007). In addition, lack of space for separate collection bins is cited as a restricting factor in the more densely populated areas. The separately collected organic waste is transported to composting facilities where compost (for sale at a price of 13 € m^{-3}) is produced[54].

Hamburg citizens pay 1.57 € m^{-3} for their drinking water plus a basic fee of about 2 € per month (costs for 2009). This is in line with the average drinking water fee in Germany of 1.85 € m^{-3} (ATT et al., 2008). On average, German households spend about 0.5% of their household income on drinking water. Sanitation costs in Hamburg are coupled to drinking water consumption and amount to 2.67 € m^{-3}, which is a bit higher than the average German fee of 2.28 € m^{-3} (ATT et al., 2008). Households that have their organic waste collected by the public cleansing service pay a fee of about 13 € per month, which is considerably less (roughly 20%-30%) than the equivalent collection of other household waste (SRH, 2009).

In 2006 Hamburg's water and wastewater utilities merged into Hamburg Wasser. In order to be prepared for future challenges and to develop export services, Hamburg Wasser pilots several innovative water and wastewater technologies. These include, for example, the feed-in of biogas into the public gas supply network (Schurig, 2009), heat recovery from wastewater sewers (Werner and Augustin, 2009), and separate collection and treatment of blackwater (Schonlau et al., 2008). Flooding from increased rainfall intensity due to climate change, as well as increasing impervious surface areas, presents a further challenge that is important in the Hamburg context. Rainwater management is however, only marginally addressed in this thesis.

3.5.2 System boundary and conceptual model

Figure 3.2 gives an overview of the specific processes and flows that are included in the analysis of Hamburg. The continous line represents the system boundary, whereas the dotted lines mark the sections in this thesis in which the respective processes are described in detail. Please note that in the figure all processes are depicted, but not every process is considered in every system. More information on the different systems is provided in Section 3.5.3.

Water bodies (surface water and groundwater) and the atmosphere are not included within the system boundaries, but exports and imports from these processes are considered. In addition, export and imports such as inputs to the households, are taken into consideration.

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[54] In 2006, a new biogas plant opened in Hamburg, which anaerobically digests industrial organic waste as well as small volumes of domestic organic waste. However, this is not considered in the system analysis, since it is a rather new development and not yet working on full-scale at the time of writing.

Industrial water supply and wastewater management are not included in the systems. On the one hand, many industrial wastewater discharges are direct discharges and it is expected that in future source separation of industrial wastewater and on-site recycling and treatment schemes will become even more prevalent[55]. On the other hand, the disregard of industrial activities, which are usually very specific, allows better generalising and comparison to other cities.

[55] In Germany 18,874 million m^3 industrial wastewater were directly discharged surface waters in 2004 (without water from thermal power plants), whereas 2,339 million m^3 were indirectly discharged into the sewer system (Statistisches Bundesamt (Destatis), 2008).

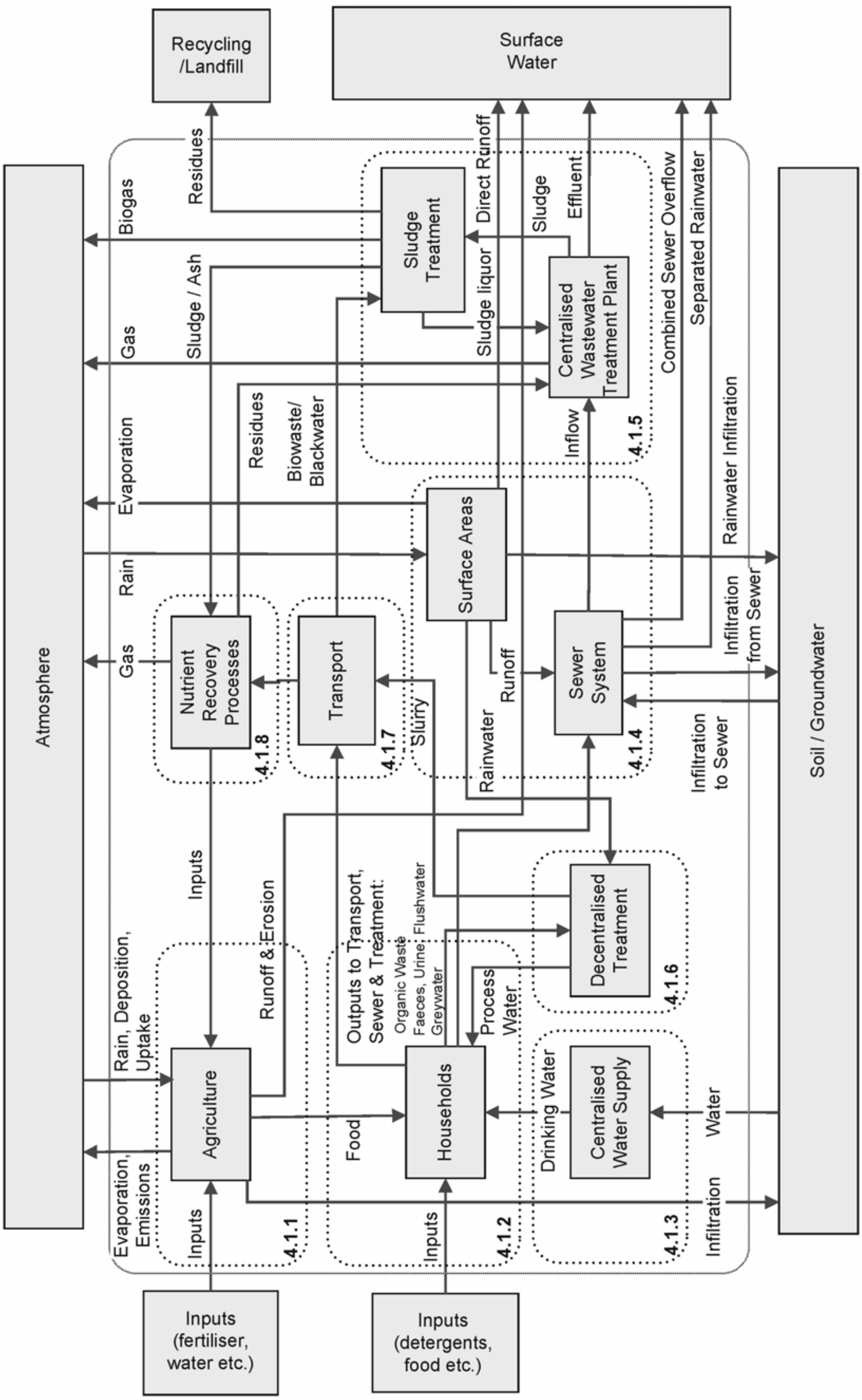

Figure 3.2 Processes and flows of the Hamburg system

(Continuous line: system boundary, dotted lines: subsystems, which are explained in Chapter 4. Numbers indicate the respective sections in this thesis.)

3.5.3 Selected systems

In this section the systems that are analysed in this study are introduced in detail. Their technical characteristics and their particular features are presented. If characteristics are not described explicitly, it is assumed that there are no differences as compared to the current situation. The current situation is included as a reference system, but in accordance with the chosen system boundaries, some processes such as industry are omitted. This omission has the effect that the results for the current situation deviate in some aspects from actual values, for example from measurements in the wastewater treatment plant.

The five systems that are investigated in this thesis are partly based on a scenario analysis initiated by the Hamburg Public Sewage Company in 2005 (Hiessl and Toussaint, 2005)[56]. This investigation differs from the scenario analysis however, with regard to the specifications and scope, since the focus is on technology in particular and not on organisational aspects or societal developments.

3.5.3.1 Current Situation (1 CurS)

As a reference for the other systems, a system called Current Situation CurS (similar to an As-Is scenario or Do-Nothing scenario) is included in the analysis. It reflects the current, conventional infrastructure in Hamburg with regard to water, wastewater and organic waste management (see Figure 3.3). It should be noted that the scheme is only a representation of the main characteristics and does not include all the considered processes.

Groundwater from Hamburg and the surrounding area is used as sole water source for the centralised water supply to the households (Leonhardt, 2005). There is no separation of domestic wastewater, which means that greywater and blackwater from

[56] The scenario analysis by Hiessl and Toussaint (2005) looked at the question of what the water infrastructure and water related service provision in Hamburg could look like in 2050. The overall aim of the analysis was the development of a strategic action plan. More than 50 external factors from sectors such as demography, technology, environment, society, economics, policy and legislation were included in the analysis, looking at their possible developments and respective impacts on the water infrastructure. About 20 persons from different authorities and institutions participated in the scenario workshops. One of their tasks included the ranking of different impact factors with regard to their forecast-uncertainty and significance. Factors that are of importance for this thesis included "resource scarcity" and "material/energy efficiency"; they both ranked among the factors with relatively low uncertainty, yet with high importance. This shows that their integration into planning for future systems is crucial. Four different scenarios or "possible futures" were developed within the framework of this scenario analysis, including some general information on technical options and processes.

conventional flush toilets are discharged in a centralised sewer system and subsequently treated in a centralised setup consisting of two combined wastewater treatment plants using activated sludge reactors. The sewer system is a mixed sewerage system for the inner part of the city, i.e. rainwater is added to the domestic wastewater. In the outer parts of the city rainwater is discharged into the environment by separate sewers. Sludge from the activated sludge reactors is anaerobically digested, dewatered, dried and incinerated. The residues (ash, gypsum) are recycled in industry and partly disposed as non-recyclable residue (heavy metal sludge) (Albaum et al., 1999).

Nutrients for agricultural production are supplied by mineral fertilisers and to some extent by organic fertilisers such as compost, but no products from the wastewater system are used. Composting of organic waste is the only nutrient recovery process included in this system.

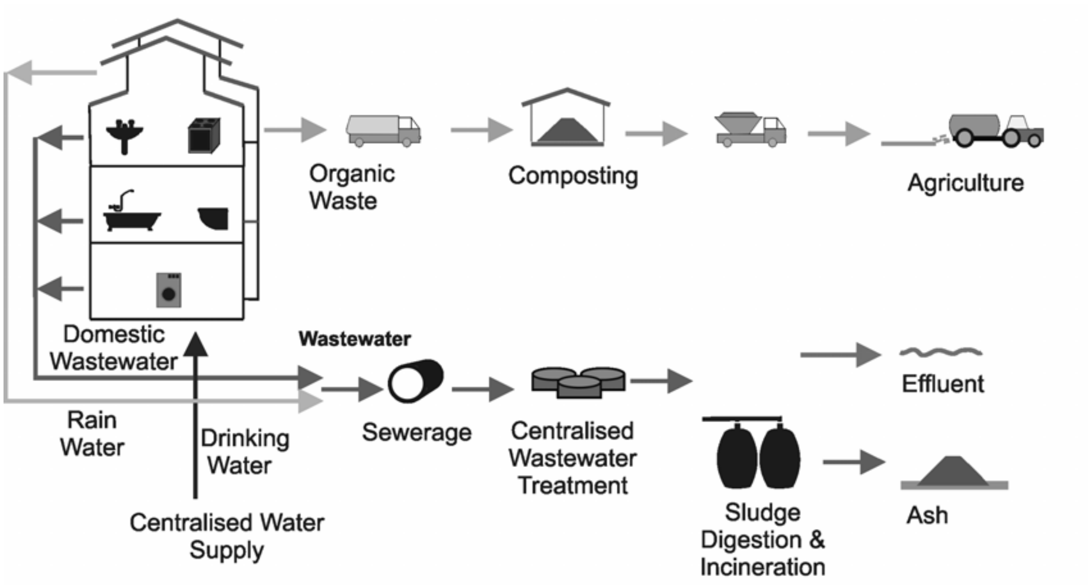

Figure 3.3: Illustration of the System Current Situation (1 CurS)

3.5.3.2 Nutrient Recovery from Sludge (2 NuRS)

The first alternative system represents nutrient recovery from wastewater in a centralised way. It is designed to be as similar as possible to the current situation and does not include any changes in household sanitary installations or in the basic water and sanitation infrastructure. Recent developments in conventional wastewater management, which focus on the recovery of nutrients from wastewater or sludge by different processes are however included (see also Section 2.4.3). The processes that can be applied depend on the wastewater treatment plant. Some of the available processes cannot be used to upgrade the current system in Hamburg, since they require, for example, biological P elimination. For this analysis, processes similar to the Seaborne® technology are selected, since the Seaborne® method is a comparatively well-established and -researched method. It has been tested since 2000 and has been in operation in a large scale treatment plant since 2007 (Montag, 2008). The final products of the Seaborne® process, i.e. struvite and ammonia solution, which are accepted fertiliser products, are used as nutrient input in agriculture.

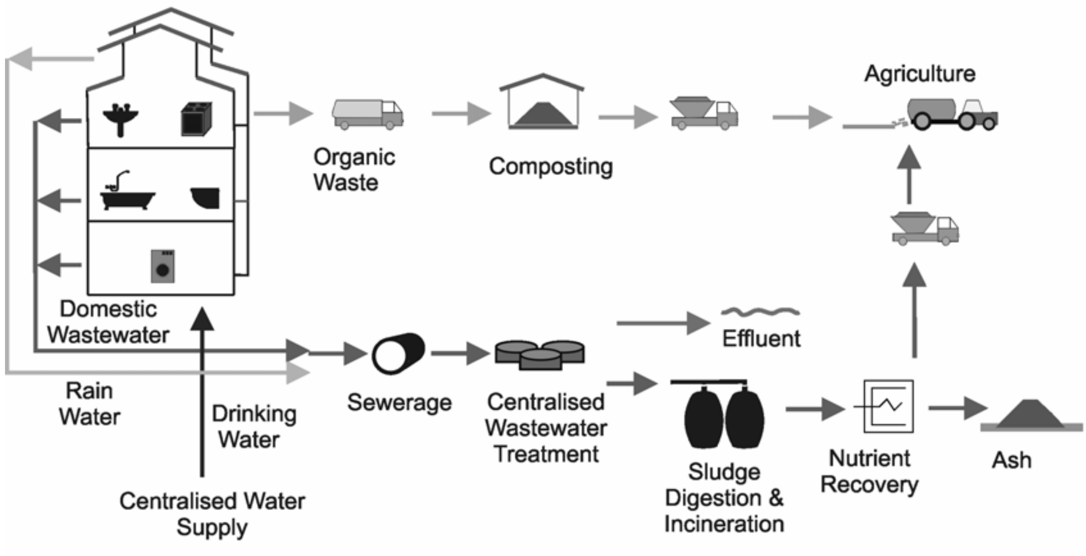

Figure 3.4: Illustration of the System Nutrient Recovery from Sludge (2 NuRS)

3.5.3.3 Nutrient Recovery from Urine (3 NuRU)

The System Nutrient Recovery from Urine (NuRU) is based on urine separation in households, institutions and public toilets. Waterless urinals and urine-diversion toilets without flush for the urine are installed. The undiluted urine is stored in the households/neighbourhoods, regularly collected by lorry and then treated in so-called nutrient-recovery stations (see Figure 3.5). It is assumed that four such stations exist within the city of Hamburg. Here, MAP is precipitated for phosphorus and nitrogen recovery. Subsequently, the remaining nitrogen is recovered by steam stripping, producing an ammonia solution. The processes for urine treatment are applied for volume reduction, elimination of harmful substances in the urine and production of accepted fertiliser products. One of the advantages of treating urine is the fact that the volume is relatively small compared to other wastewater flows. MAP and the ammonia solution are transported to agriculture. To improve biogas production the collected organic waste is co-digested in the sludge treatment plant of the centralised wastewater treatment plant.

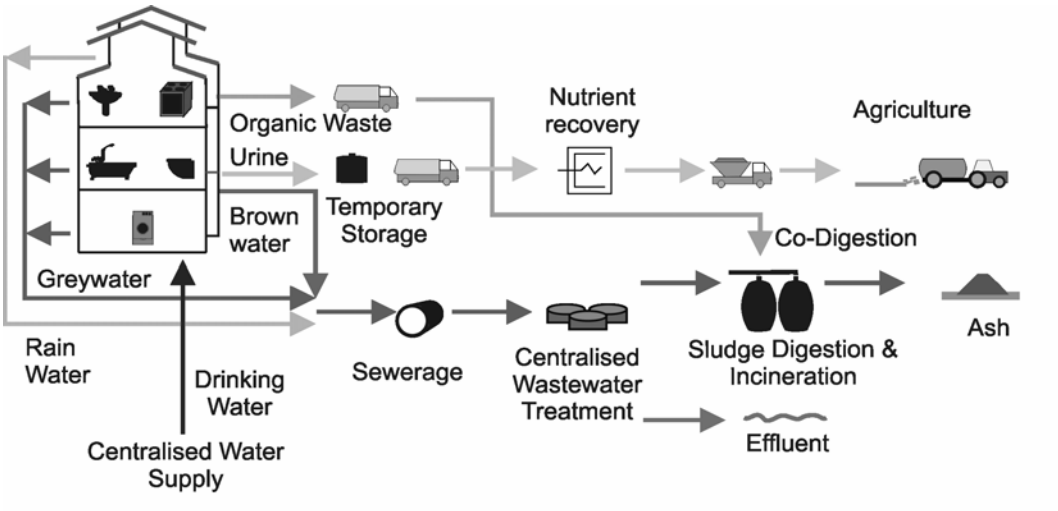

Figure 3.5: Illustration of the System Nutrient Recovery from Urine (3 NuRU)

3.5.3.4 Co-Digestion of Blackwater and Organic Waste (4 CoDig)

The centralised wastewater system is adapted in such a way, that only greywater is transported in the conventional sewer system. Blackwater is collected in vacuum toilets with relatively low dilution, and then transported to temporary storage tanks via a vacuum sewer system. A lorry-based collection system is in place to transport blackwater to the centralised sludge treatment plant, where it is co-digested with organic waste and excess sludge from the aerobic greywater treatment. Anaerobic treatment of blackwater has the advantage that the full energetic potential can be utilised, instead of using energy for aerobic treatment. Sludge liquor, i.e. water from dewatering sludge, is treated by steam stripping to recover ammonia. Phosphorus is recovered from the ash resulting from sewage sludge incineration. This recovery process is based on the BioCon® process (see section 2.4.3). For a schematic view of the system please refer to Figure 3.6.

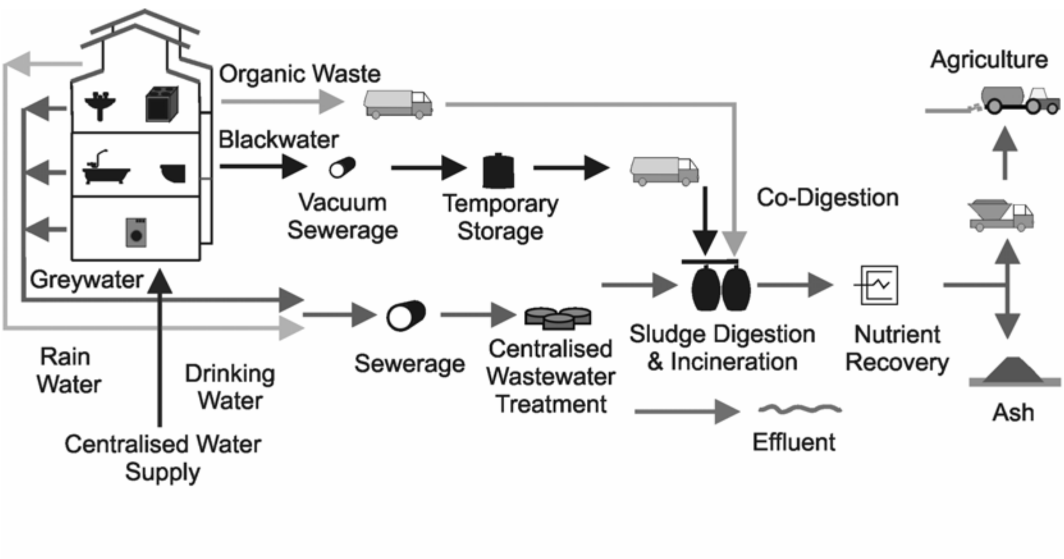

Figure 3.6: Illustration of the System Co-Digestion of Blackwater (4 CoDig)

3.5.3.5 Blackwater digestion in decentralised facilities (5 BlaD)

The System Blackwater Digestion (BlaD) (see Figure 3.7) includes the concept of neighbourhood treatment plants. This makes the centralised water and wastewater system redundant and allows a higher degree of flexibility and adaptability. Concentrated blackwater from vacuum toilets is transported via vacuum sewers to anaerobic digesters (biogas plants) that serve a cluster of houses with about 3,000 to 5,000 inhabitants. Organic waste is shredded and added to the biogas plants. The sanitised and digested slurry is collected by lorries, brought to one of four storage stations and used as fertiliser in agriculture. Therefore, no additional treatment processes are required for nutrient recovery, but volumes to be transported are relatively high. Greywater is treated by pre-treatment and aerobic treatment, such as in decentralised membrane bioreactors (MBR), which is a combination of activated sludge treatment and filtration processes (i.e. ultrafiltration). The quality of the treated greywater is high enough to allow the use of greywater as process water in the households. Sludge from the MBR is added to the biogas reactors. Bottled water is used as drinking water source. Rainwater is locally infiltrated, but also partly used to replenish process water. Wherever decentralised rainwater management is not possible, rainwater sewers are used to discharge the rainwater into the environment.

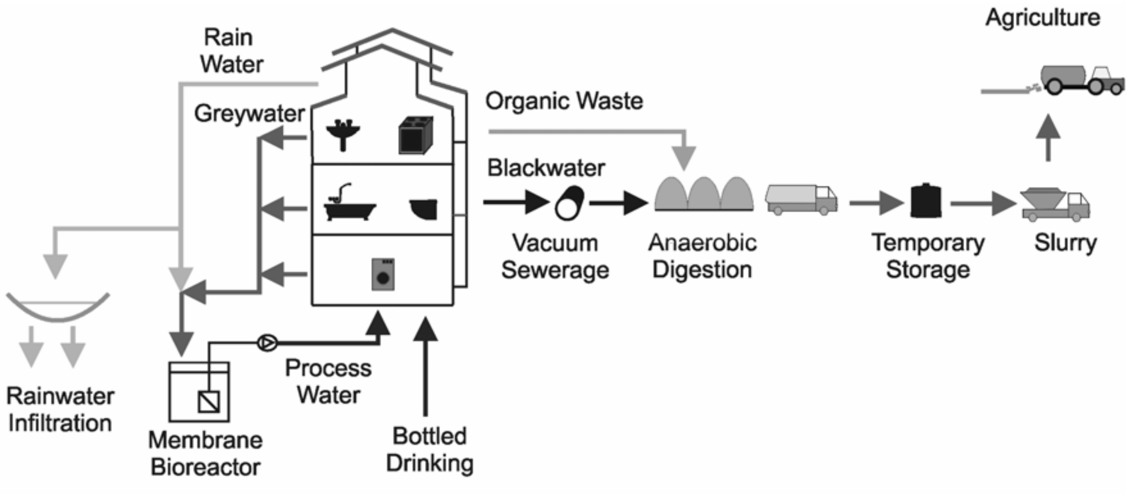

Figure 3.7: Illustration of the System Blackwater Digestion (5 BlaD)

3.5.3.6 Composting of faeces and use of urine (6 CompU)

The System Composting of Faeces and Use of Urine (CompU) is characterised by source separation of all wastewater fractions and their decentralised treatment (see Figure 3.8). Urine and faeces are separately collected in dry (waterless) toilets (UDDT). Both fractions are stored on-site until they are picked up in a lorry-based collection system. After collection the urine is simply stored in one of four storage stations. Subsequently the urine is used in agriculture. Faeces are first dried on site and then collected and composted together with organic waste; this is to reduce the pathogen content, before the compost is used in agriculture. Some authors recommend composting toilets or containers underneath the toilets (Berger, 2002; Del Porto and Steinfeld, 1999; Jenkins, 2005). However, this requires regular maintenance activities by the households (e.g. turning of compost heap), which cannot be expected on a large scale. Therefore, a combination of drying/storage on site and collection and treatment in a centralised plant is suggested in this system. Greywater is separately collected and treated in on-site membrane bioreactors (MBR), including disinfection by UV for subsequent use as drinking and process waters in the households. Residues from MBR treatment are added to the composting processes. Rainwater is locally infiltrated into the soil or discharged in rainwater sewers.

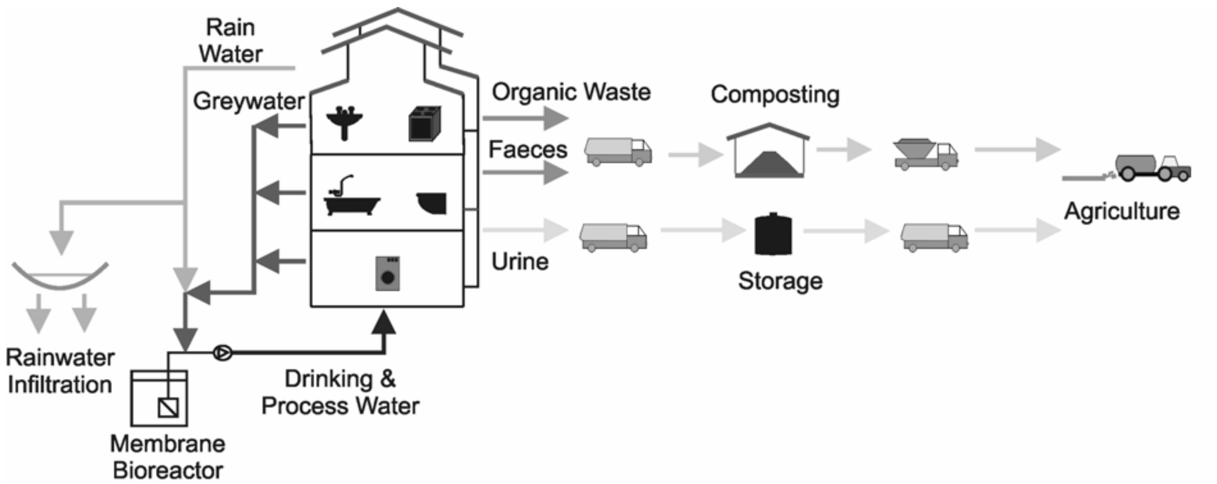

Figure 3.8: Illustration of the System Composting of Faeces and Use of Urine (6 CompU)

3.5.3.7 Overview of the selected systems

The different systems are defined in such a way that varying degrees of source separation, as well as different levels of decentralisation are represented. Table 3.1 illustrates a recapitulation of the main characteristics of the systems and shows the collection and treatment of the flows considered. System 1 CurS is not included, but basically resembles System 2 NuRS without any nutrient recovery processes. The boxes shaded in grey highlight the source separated flows. In summary, the three Systems 2 NuRS, 3 NuRU and 4 CoDig rely on centralised water and wastewater infrastructure, whereas Systems 5 BlaD and 6 CompU include decentralised neighbourhood or on-site technologies. This is also reflected in Figure 3.9, which highlights the degree of centralisation and source separation of the six systems.

Table 3.1: Overview of treatment processes in the selected systems (Hamburg)

<table>
<tr><th></th><th>2 NuRS</th><th>3 NuRU</th><th>4 CoDig</th><th>5 BlaD</th><th>6 CompU</th></tr>
<tr><td>Urine</td><td rowspan="4">To central wastewater treatment plant (WWTP), centralised nutrient recovery from sludge (Seaborne®)</td><td>Separation, transport to central nutrient recovery (MAP and NH_3 stripping)</td><td rowspan="2">Separation in vacuum toilets, transport of blackwater to central WWTP. Co-digestion and recovery of nutrients from sludge liquor (MAP and NH_3 stripping)</td><td rowspan="2">Separation in vacuum toilets, anaerobic digestion in neighbour-hood plants, transport and use of slurry</td><td>Separation, transport, storage and use</td></tr>
<tr><td>Faeces</td><td rowspan="3">To central WWTP</td><td>Dry toilets, dehydration, composting, use of compost</td></tr>
<tr><td>Greywater</td><td rowspan="2">To central WWTP</td><td rowspan="2">On-site treatment (MBR) and use as process water</td><td rowspan="2">On-site treatment (MBR) and use as drinking and process water</td></tr>
<tr><td>Rainwater</td></tr>
<tr><td>Organic waste</td><td>Composting</td><td>Co-digestion in WWTP</td><td>Co-digestion in WWTP</td><td>Anaerobic digestion with blackwater</td><td>Composting together with faeces</td></tr>
<tr><td>Water supply</td><td>Central water supply</td><td>Central water supply</td><td>Central water supply</td><td>Grey- and rainwater, supplemented by bottled water</td><td>Grey- and rainwater</td></tr>
<tr><td>Fertiliser products</td><td>MAP</td><td>MAP, NH_3 solution</td><td>MAP, NH_3 solution</td><td>Digested slurry</td><td>Stored urine, compost</td></tr>
</table>

Notes:
Systems 2, 3, 4: The central WWTP is an activated sludge plant with removal of organic matter, N and P including sludge treatment.
Systems 2, 3, 4: Only rainwater in areas with mixed sewerage is treated in the WWTP. In other areas it is discharged via rainwater sewers.

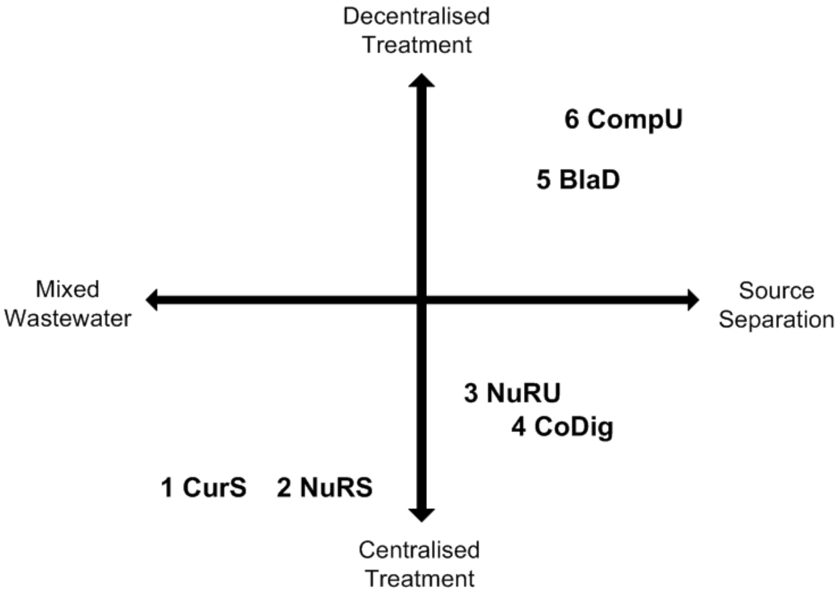

Figure 3.9: Classification of the Hamburg systems regarding their degree of centralisation and source separation (diagram adapted from Prager, 2002)

3.6 Case study Arba Minch

Water, waste and wastewater conditions and practices in Arba Minch, which is the second case study of this thesis, differ significantly from the conditions and practices in Hamburg. Therefore, the next section gives an overview of the town with a special focus on topics considered to be relevant for this study. This is followed by a definition of system boundaries and an introduction of the investigated systems.

3.6.1 Introduction to Arba Minch

Arba Minch is located in the south of Ethiopia in the Southern Nations, Nationalities and People's Regional State (about 6°2′ N and 33°3' E) (Figure 3.10). It is situated about 500 km from the capital, Addis Ababa and about 250 km from the regional capital, Awassa. About 40 years ago there were only about 2000 people settled in the Arba Minch area. Nowadays, due to migration, Arba Minch is one of the fastest growing towns in Ethiopia (Aregu and Demeke, 2006). According to CSA (2006) the population in 2005 was about 72,500. Current population estimates indicate that there are about 80,000 inhabitants in Arba Minch (AMU and ARB, 2007). The average household size is between 4.5 and 5 persons. The administrative border encloses an area of about 21.9 km^2. Thus, the average population density is about 36 p ha^{-1}. But when referring to residential areas only, the density is estimated to be about 154 p ha^{-1}; with a projected growth to up to 250 p ha^{-1} over the coming decades (DHV Consultants, 2002).

Arba Minch is located in the Rift Valley and the climate is characterised by two rainy seasons; from April to May and from September to October. The mean annual rainfall is about 890 mm y^{-1} (NMASZ, 2006), but rainfall is very erratic and in general, declines in annual rainfall due to climate change are observed (Seleshi and Zanke, 2004). Average air temperatures vary between 17°C and 30°C with an annual average temperature reported to be about 20°C (DHV Consultants, 2002). Arba Minch is characterised by its location close to two large lakes of the Rift Valley, namely Lake Abaya and Lake Chamo. Another landmark is Kulfo River, a perennial river with a mean annual flow of about 184 million m^3. The river water is not only used for irrigation, but also partly for domestic water use such as washing laundry or car washing. River pollution is a growing concern (AMU and ARB, 2007).

Figure 3.10: Location of Arba Minch
(Based on maps from NordNordWest and Alvaro1984 18, Wikimedia Commons)

Ethiopia's economy is largely characterised by agricultural activities, which account for half of the GDP and 80% of employment (UNICEF, 2005). This is however not the case for Arba Minch, where agriculture plays only a minor role in the local economy (AMU and ARB, 2007). Only few inhabitants are working as subsistence or small-scale farmers, mainly in the surrounding area of Arba Minch. On the other hand, there is a large state farm, which has been recently transferred to private investors and which covers an area of 800 to 900 ha. In addition, the university, private entrepreneurs, as well as micro and small enterprises cultivate an area of 100 to 150 ha. A detailed analysis of the agricultural stakeholders in Arba Minch and their role for resource oriented sanitation, is given by Plückers (2009).

Drinking water supply in Arba Minch, which literally means forty springs, is from groundwater sources. Annually, about 700,000 m^3 y^{-1} are extracted. Water treatment, which is basically only chlorination, and distribution were extended in 1987, and currently comprises a network of 45 km of pipes. Yet, only about 5000 households have private water taps[57], whereas the remaining households rely on water from one of the 34 public stand pipes or from private vendors (AMU and ARB, 2007). Per capita consumption therefore varies depending on the type of water supply and is calculated to be about $40\ l\ p^{-1}\ d^{-1}$ for private house connections and $5\ l\ p^{-1}\ d^{-1}$ for public stand pipes (based on AMU and ARB (2007) and survey data from the ROSA project). Since a large part of the equipment has reached the end of its economic life, and to address the population increase, the water supply scheme is currently being prepared for an upgrade, financed with support from the Ministry of Water Resources.

Sanitation in Arba Minch is basically on-site sanitation, since there is no sewer system. Only the university, with about 7000 students and staff, is connected to a pond system. In addition, a few so-called condominium houses, as well as institutions, hotels and restaurants are equipped with (pour-)flush toilets and septic tanks. By contrast, the vast majority of households (about 76%) use pit latrines, which are of low quality construction and are often subject to flooding or collapse. About 10-16% of the inhabitants have to resort to open defecation (based on AMU and ARB (2007) and the ROSA Demand Assessment, unpublished project document). Greywater[58] is usually disposed of by spilling it in the compound or onto the street (Teklemariam, 2009). Hygiene is at risk not only due to open defecation and disposal of wastewater, but also due to the lack of treatment facilities. Only in the beginning of 2010 treatment facilities have been constructed[59].

Another problem in this regard is the unresolved institutional responsibility for sanitation and wastewater management. Not only the Town Water Service, which is also sometimes referred to as Water Supply and Sewerage Enterprise, but also the municipality, which has recently established a sanitation department, have legal authorisation for dealing with sanitation issues. But a clear allocation of responsibility with regard to tasks has not yet been decided, and the activities of both parties have been rather marginal up to now. At this stage, therefore, sanitation is more of private than of public concern. Nevertheless, the recent activities in Arba Minch within the

[57] Only a small fraction (i.e. 3%) has an in-house connection, and the majority uses yard taps.

[58] Teklemariam (2009) analysed greywater samples in Arba Minch and could show a largely increased pollutant load compared to European data. The reason for high concentrations is supposed to be multiple uses of the water and low water consumption.

[59] Up to the middle of 2009 the town did not have its own vacuum truck for the collection of slurry from pits and septic tanks; when needed a truck had to be hired from distant towns at great expense.

framework of the EU-project ROSA, sparked initiatives in various stakeholders to set sanitation higher on the agenda.

Analogue to sanitation, solid waste management is rather a household responsibility than a public responsibility. According to AMU and ARB (2007) only about 5 to 10% of the households are serviced by waste collectors, who dump the waste on open disposal sites. The remaining population either dumps the waste in pits or on open fields, or burns it. Except for cow manure, organic waste has generally not been considered for reuse. An initiative started by the ROSA project to organise composting activities, is now carried out on a small scale by private enterprise (Ercolano, 2009).

The cost of drinking water depends on the type of supply. The fee for house connections is related to average consumption and is 1.5 birr m^{-3} for a monthly consumption of up to 10 m^3 and 2.1 birr m^{-3} for a monthly consumption of up to 30 m^3. From public taps the water is sold at 3.33 birr m^{-3} and vendors sell it at 7.5 birr m^{-3}. Calculations from the data gathered during the ROSA Demand Assessment, shows that the average household cost for water is about 15 birr per month, which is about 2.7% of the average household income of 560 birr per month[60]. Teklemariam (2009), who did a study on greywater in Arba Minch, showed that as much as 5-10% of the household income is spent on water. Connection fees for household connections of 400 to 750 birr are often considered too expensive, so that many households cannot afford to get connected (AMU and ARB, 2007). As discussed above, up to now public utilities have not delivered any sanitation services and therefore there is no sanitation fee. Even so, households pay daily labourers for empting or shifting their pit latrines or VIP, and users of septic tanks need to hire vacuum trucks for emptying. Costs for these services depend on the required frequency. Pit emptying costs about 50 to 100 birr, whereas the digging of a new pit is in the range of 15 to 50 birr per meter depth (AMU and ARB, 2007). Micro and small enterprises usually charge the households for solid waste collection on an affordability basis.

Apart from the challenges with regard to water supply and sanitation, Arba Minch faces other environmental problems such as uncontrolled waste disposal and deforestation. According to Aregue and Demeke (2006) deforestation because of fuel wood collection and timber extraction, is a major threat for Arba Minch, resulting in land erosion and eventually leading to micro-climate changes and a decrease of water availability. Therefore, the generation of energy from alternative fuel sources such as biogas, is an important issue.

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[60] Please note that the data from the demand assessment survey is showing a high variability and uncertainty and should therefore be considered with care.

3.6.2 System boundary and conceptual model

The system boundary and the processes that are included in the system analysis are depicted in Figure 3.11. Again, the continous line represents the system boundary, whereas the dotted lines group the processes as they are described in the respective sections of Chapter 5.1. The figure illustrates all the flows and processes that are included in the model (as mentioned in Section 3.3), but it should be noted that not all of them are applicable in all systems.

Institutions such as schools or hospitals are not explicitly included in the model. However, for the case of Arba Minch the university is included as process, because students and staff residing on campus make up about 10% of the Arba Minch population and the university has its own wastewater management system. Besides a textile industry and some small companies, industry does not play a major role in Arba Minch. The wastewater management of industry is managed separately from domestic waste(water) and is not considered for water reuse or nutrient recovery. Therefore, industry is not included in the analysis.

In contrast to the case study Hamburg, where livestock and the management of manure is not considered, the Arba Minch model includes the process Livestock. This is because, firstly, livestock plays a more important role for Arba Minch than for Hamburg and, secondly, manure is currently not managed or properly disposed of in Arba Minch. Therefore, it is expected that synergies may be developed by a combination of treating human waste and animal waste.

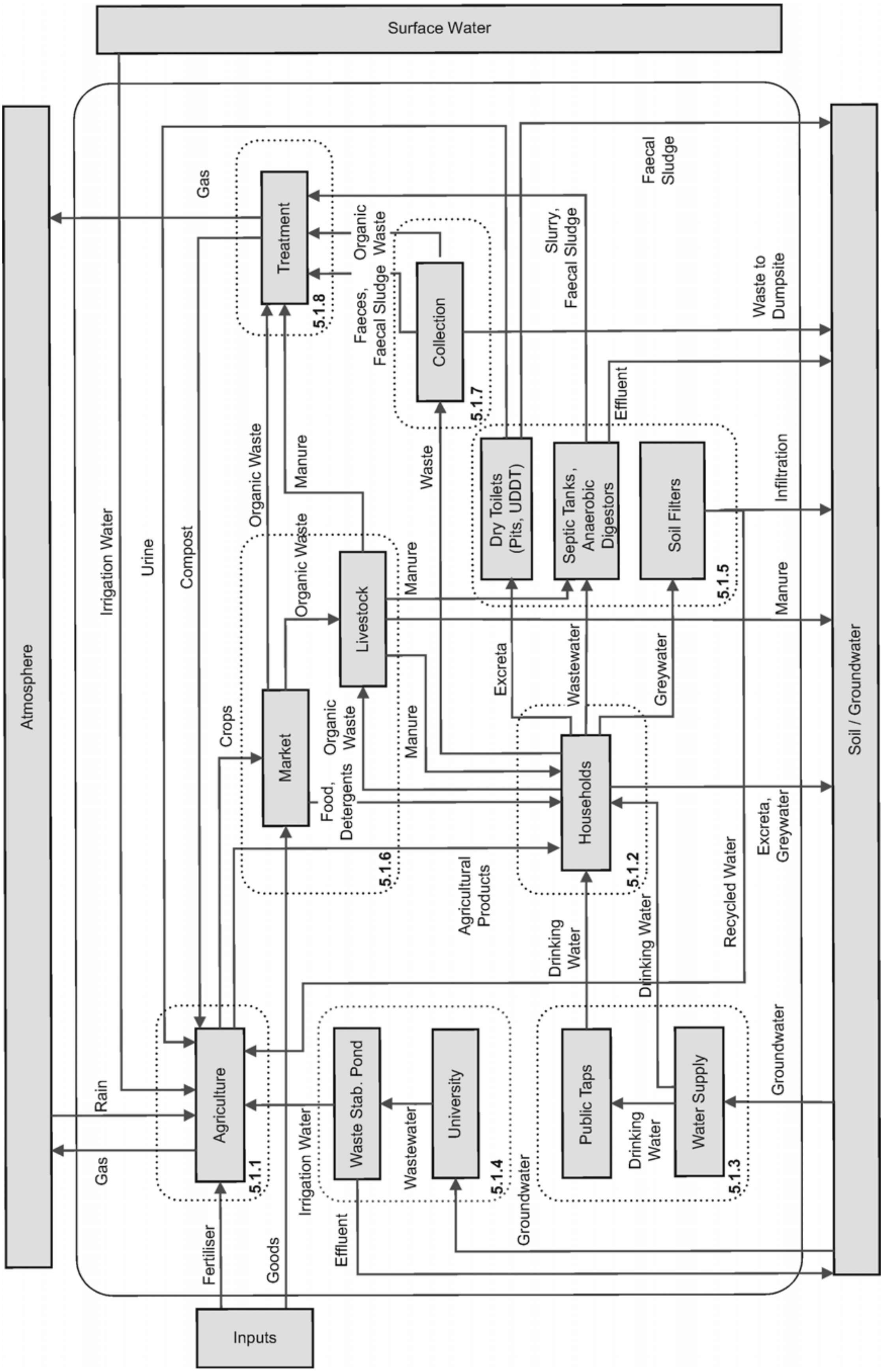

Figure 3.11: Processes and flows of the Arba Minch system
(Continous line: system boundary, dotted lines: subsystems, which are explained in Chapter 5. Numbers indicate the respective sections in this thesis.)

3.6.3 Selected systems

For the case of Arba Minch three alternative systems, which represent three different ways of closing the loop between sanitation and agriculture, are selected. In addition, the current situation is included as reference system. In the following sections the four investigated systems are briefly introduced. A detailed description of the relevant processes follows in Section 5.1.

3.6.3.1 Current Situation (1 CuSit)

In the reference system (the current situation) the majority of the households use pit latrines as a sanitation facility. Some households and institutions have septic tanks (about 6%) and about 16% of the population practice open defecation. Faecal sludge from septic tanks and pit latrines is either emptied by vacuum truck or manually (see Figure 3.12). There is no reuse, but the sludge is disposed of in the environment. Some of the domestic waste is collected and dumped; organic waste is not recycled. The university has its own sanitation system, which is water-based and discharges the wastewater to a series of wastewater stabilisation ponds. Industry (e.g. textile industry) is not included in the analysis.

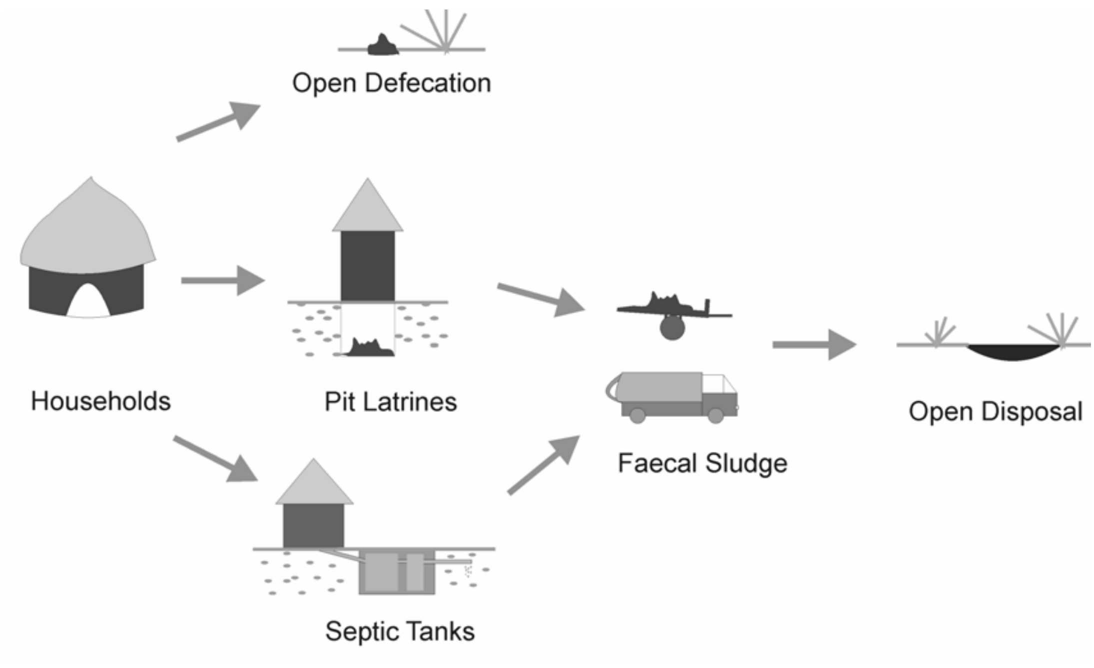

Figure 3.12: Illustration of the System Current Situation (1 CuSit)

3.6.3.2 Co-Composting of Faecal Sludge and Organic Waste (2 CoComp)

The System Co-Composting of Faecal Sludge (Figure 3.13) is based on sanitation facilities as in the current situation, which means that there are no changes in the sanitary installations of the households. Treatment facilities for co-composting of the collected faecal sludge and separately collected organic waste are however included. The compost is used in agriculture as soil conditioner.

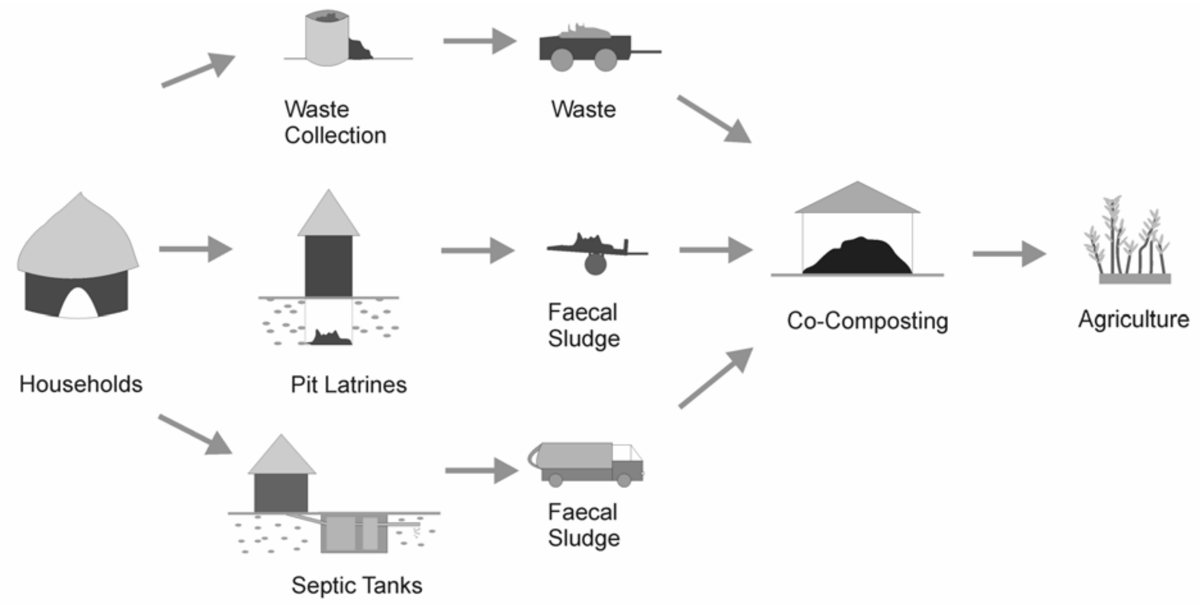

Figure 3.13: Illustration of the System Co-Composting of Faecal Sludge and Organic Waste (2 CoComp)

3.6.3.3 Urine-diverting, Dry Toilets (3 UDDT)

The System Urine-Diverting, Dry Toilets (UDDT) is based on the introduction of source-separating toilets (Figure 3.14). Urine is separately collected, which also has the advantage that smell and flies are reduced due to the lowered moisture content of the faecal matter as compared to the faecal matter in a conventional pit latrine. In addition, basic UDDT require no water for flushing. Urine is stored on site until it is collected and used as a nutrient-rich supplement in agriculture. Additional storage near agricultural lands is required to allow further pathogen die-off and to ensure that the urine-based fertiliser is only applied when needed by the plants. Faeces are collected in the toilets together with drying additives such as soil or ash. After a sufficient storage and dehydration time on site, the faeces are collected and co-composted together with organic waste in centralised facilities. Faecal sludge from septic tanks can be added to the co-composting process. Greywater is either locally infiltrated into the ground if quantity and quality allows, or it is treated in planted soil filters or greywater towers. Thus, the treated water can be provided as irrigation water for on-site (subsistence) farming.

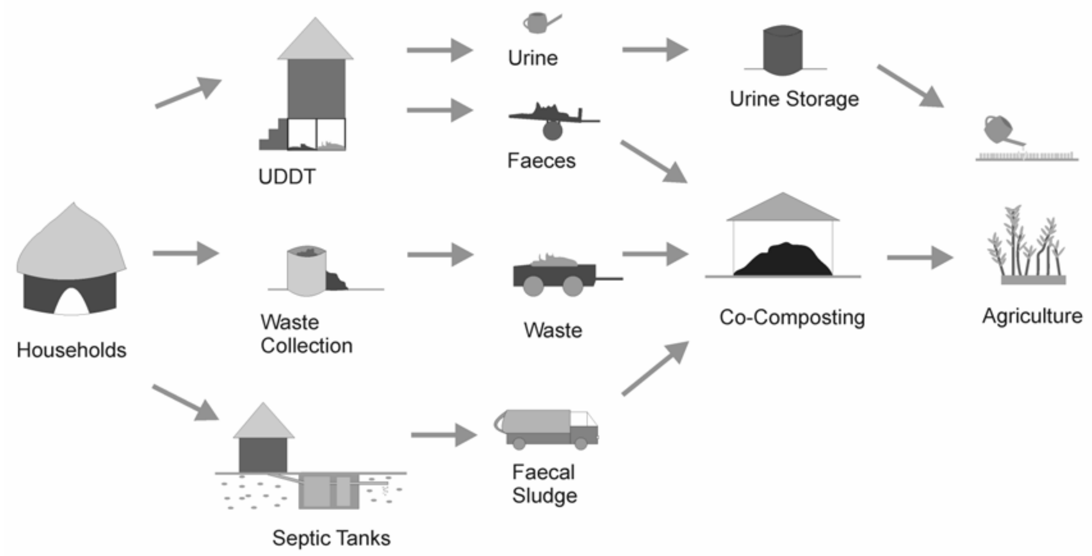

Figure 3.14: Illustration of the System Urine-Diverting, Dry Toilets (3 UDDT)

3.6.3.4 Anaerobic Digestion (4 AnDig)

In System 4 AnDig the wastewater from pourflush toilets is treated in anaerobic digesters, to which clusters of households are connected (Figure 3.15). Animal manure and organic waste are added to the digestion process; this allows a combined treatment of different waste flows. The digested slurry is composted and subsequently used in agriculture as fertiliser. The gas that is produced in the anaerobic process is used by the households for purposes such as cooking or lighting. Greywater is either locally infiltrated if quantity and quality allows, or treated in planted soil filters or greywater towers. The treated water can thus be used as irrigation water for on-site (subsistence) farming.

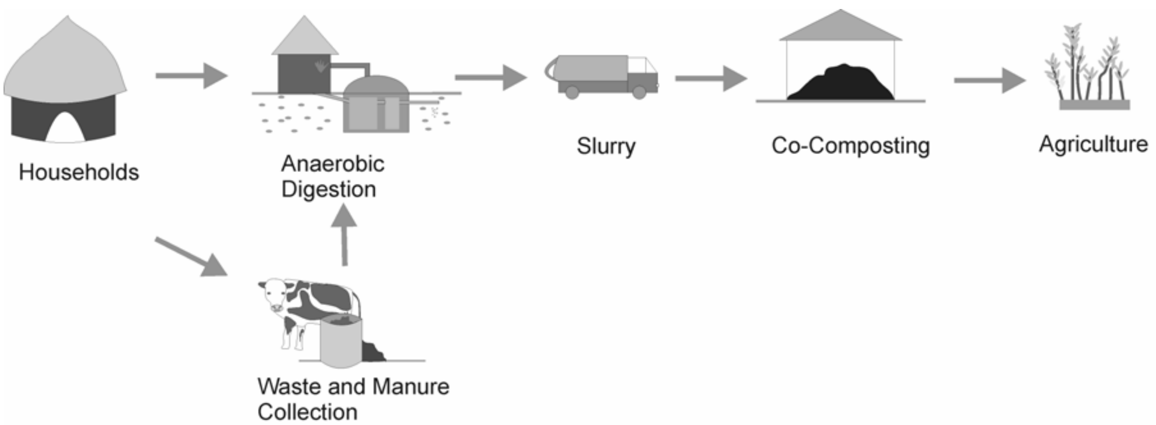

Figure 3.15: Illustration of the System Anaerobic Digestion (4 AnDig)

3.6.3.5 Overview of the selected systems

Table 3.2 and Figure 3.16 give an overview of the alternative systems selected for the case study Arba Minch. The characterisation of the systems regarding their degree of centralisation is rather ambiguous, since there is always a mixture of decentralised facilities (e.g. storage of waste flows on-site) and centralised facilities (e.g. centralised storage or treatment units). Yet, the figure shows a rough classification regarding the main treatment steps.

Table 3.2: Overview of treatment processes in the selected systems 2-4 (Arba Minch)

<table>
<tr><th></th><th>2 CoComp</th><th>3 UDDT</th><th>4 AnDig</th></tr>
<tr><td>Urine</td><td rowspan="3">Faecal sludge from pit latrines and septic tanks added to co-composting</td><td>Separation, transport to storage and use</td><td rowspan="2">Anaerobic digestion</td></tr>
<tr><td>Faeces</td><td>Dehydration, composting</td></tr>
<tr><td>Greywater</td><td>Low-cost treatment (soil filters)</td><td>Low-cost treatment (soil filters)</td></tr>
<tr><td>Rainwater</td><td>No collection</td><td>No collection</td><td>No collection</td></tr>
<tr><td>Organic Waste</td><td>Co-composting with faecal sludge</td><td>Co-composting with faeces</td><td>Anaerobic digestion</td></tr>
<tr><td>Water Supply</td><td>Central water supply</td><td>Central water supply & recycled greywater for agriculture</td><td>Central water supply & recycled greywater for agriculture</td></tr>
<tr><td>Fertiliser products</td><td>Compost</td><td>Urine, compost</td><td>Digested and composted slurry</td></tr>
</table>

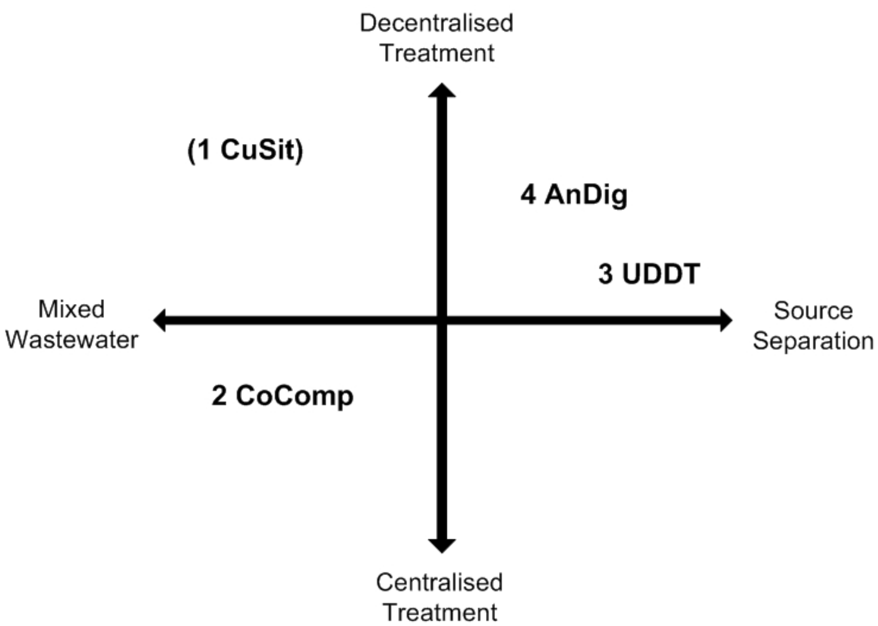

Figure 3.16: Classification of the Arba Minch systems regarding their degree of centralisation and source separation (diagram adapted from Prager, 2002)

3.6.3.6 System parameters for the Arba Minch ceMFA model

By contrast to the Hamburg case study, the Arba Minch systems are integrated into one equation system. Therefore, system parameters are introduced that define the characteristics of the four systems (see Table 3.3). These system parameters can be varied, for example, to analyse the effect of only a partial implementation of particular system components. It is assumed that open defecation will be eradicated in Systems 3 UDDT and 4 AnDig and instead those households will use either UDDT or pour-flush toilets with anaerobic treatment. The percentage of households with septic tank systems in systems 2 CoComp and 3 UDDT is assumed to be the same as in the current situation.

Table 3.3: System parameters used for defining the Arba Minch systems (implementation rate in %)

	1 CuSit	2 CoComp	3 UDDT	4 AnDig
type of toilet facility				
open defecation	16	16	0	0
pour-flush toilet with anaerobic treatment	6	6	6	100
of which biogas plants are	0	0	0	100
UDDT	0	0	94	0
pit latrines	78	78	0	0
organic waste from households				
given to livestock	30	30	30	30
collected	6	60	60	60
collected and added to co-composting	0	100	100	0
collected and added to biogas plants	0	0	0	100
organic waste from market				
added to co-composting	0	100	100	100
manure				
used as fuel	10	10	10	10
added to co-composting	0	20	20	0
added to biogas plants	0	0	0	20
greywater recycling	0	0	50	50

4 Hamburg

This chapter presents the ceMFA of the case study Hamburg. In Section 4.1 the processes that are included in the model are described in detail. Section 4.2 contains the main results of the modelling, followed by the presentation of sensitivity analyses and parameter variations in Section 4.3. The chapter concludes with a discussion of possible transformation processes.

4.1 Process descriptions

In the following sections the different processes, that are included in the different systems and their inflows and outflows, are characterised. The parameters used for modelling mass and nutrient flows, as well as energy and costs, are explained. Those parameters, which are not included in the respective sections, can be found in Annex A.

4.1.1 Agriculture

Closing the loop of the wastewater system means to recycle nutrients from human waste products as plant fertiliser for their use in agriculture. Therefore, agriculture is considered as an important process in the system analysis. The flows that are connected to the process Agriculture are depicted in Figure 4.1. Of particular importance is the flow Food connected to the process Households, as well as the two flows called Inputs, which represent the provision of fertiliser as mineral fertiliser from outside the system boundary and as recycled product from within the system respectively.

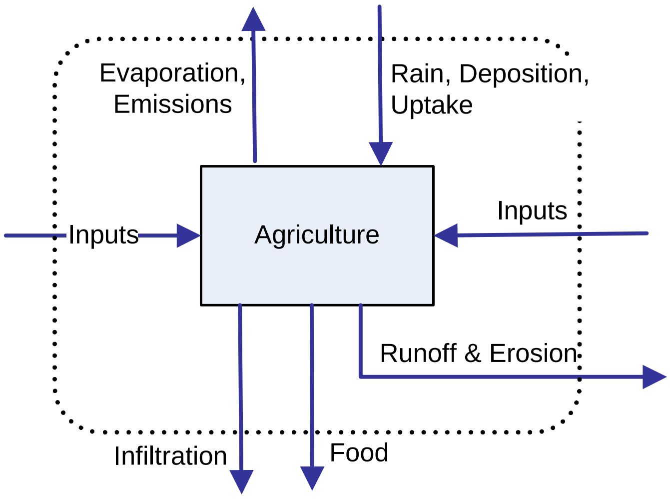

Figure 4.1: Process Agriculture

Agricultural production is a very complex process, since products and cultivation methods vary and depend on local conditions. In order to simplify the analysis, the following assumptions are taken into consideration:

- Fertiliser requirements depend strongly on the kind of crop, crop rotation, soil type and available nutrients in the soil. For simplification, average fertiliser requirements are assumed.
- It is assumed that animal husbandry practices do not vary in the different systems, which leads to following conclusions:
  - Emissions from animal husbandry, which can contribute significantly to ammonia emissions, are neglected, because they are the same in all systems.
  - Application of livestock manure as fertiliser is not considered, because it contributes constant amounts of nutrients to the soil. In addition, because the system analysis is limited to domestic wastewater, the anaerobic digestion of animal manure for energy production is not included.

Mass and nutrient flows

Since Hamburg is a city state and has therefore only limited agricultural areas, the system boundary is extended to include an overall agricultural area large enough to theoretically feed the population of Hamburg (i.e. including a concept of "hinterland"). Baccini and Brunner (1991) state that on average about 4m^2 of farming land are needed to produce 1 kg of food. This is in line with Schmid Neset (2005) who concludes that about 0.25 ±0.05 ha[61] per person are necessary to produce the average Swedish food consumption of 639 kg cap^{-1} y^{-1}, if all the food (including fodder for livestock) would be produced locally. Thus, the total agricultural area considered in the analysis consists of

[61] In Germany, about 17million ha land is used for farming (Wikipedia, 2009). This is equivalent to about 0.2 ha per capita.

the agricultural area in Hamburg, which is 19,189 ha (Statistisches Amt für Hamburg und Schleswig-Holstein, 2004), plus a surrounding hinterland area of 487,497 ha[62]. The hinterland is calculated in such a way that the amount of nationally produced food required for Hamburg's population can be cultivated on the total area assuming 4 m^2 per kg food. Imported food is not considered. Regarding the amount of consumed food and its origin, the FAO Food Balance Sheet Germany (FAO, 2004) is used to calculate the total amount (about 726 kg $cap^{-1} y^{-1}$), the ratio of imported food and the respective nutrient contents (see Annex A).

The fertiliser inputs are calculated in such a way that for every nutrient an average fertiliser requirement per hectare is assumed. No differentiation regarding the specific crop requirements is made, but averages of mineral fertiliser consumption in Germany for the year 2006/2007 are used (see Table 4.1). Other fertilisers like farm fertilisers or secondary raw material fertilisers are not considered, but assumed to be constant in all systems since they are supply-based and not dependent on demand. The model calculates the required mineral fertiliser input based on the total fertiliser demand (i.e. the current mineral fertiliser input) minus the amount of nutrients that are recycled from human waste products in Systems 2 to 6. It is assumed that the fertilising effect of nutrients from source separated flows is comparable to that of mineral fertiliser[63]. Ammonia volatilisation after fertilisation with urine can amount to up to 10% (Muskolus, 2008; Kirchmann and Petterson, 1995) (see also Section 4.1.8.5), but this is comparable to the gaseous nitrogen emissions of mineral fertilisers, which are set in the model to be 10±5% (Stroh and Djeradi, 2007). These losses are considered as gaseous nitrogen outputs. In compost, the immediate plant availability of nutrients is relatively low[64], but since the compost contributes to long term soil fertility the nutrient input to agriculture is set in the model to be fully effective.

Table 4.1: Average mineral fertiliser application rates for Germany

Nutrient	**N**	**P**	**K**	**S**
[kg ha^{-1} y^{-1}]	94±9[a,b]	7±1[a,b]	21±2[a,b]	20±8[c]

References: a) Statistisches Bundesamt, 2007, b) IFA, 2008, c) Kemira GrowHow GmbH, 2006

[62] This represents about 48% of the agricultural area of Schleswig-Holstein (MLUR, 2005) or 18% or the agricultural area of Niedersachsen (LSKN, 1999).

[63] Tidaker et al. (2007) mention that yield decreases can occur with urine application as a result of soil compaction (1.5%), wheel traffic in growing crops (2%) and higher ammonia losses (2.4%). However, these aspects are neglected in this study.

[64] Muskolus (2008) reports that less than 5% of the compost nitrogen is immediately plant-available.

The mass balance is implicating a water balance for the flows Rain, Evaporation and Irrigation Inputs. Furthermore, the flows Runoff, Erosion and Infiltration, which are important for the nutrient balances, are calculated using data from the Elbe Watershed Management and Monitoring Programme (Behrendt et al., 2003) and adapted to the hinterland. Atmospheric uptake and biological fixation of nitrogen are assumed to be in the same order as the ammonia-nitrogen emissions by plants and therefore these two flows are set to zero (Arman et al., 2002). Sulphur is atmospherically deposited although this value has decreased in recent years due to mitigation of acid rain. Emissions, particularly of nitrogen during fertilisation, and factors indicating the plant availability are however included as parameters.

Energy

Energy consumption for production of mineral fertiliser contributes significantly to the overall energy demand and is therefore included in the system analysis. The values for the specific primary energy consumption of fertiliser provision used in this study, are based on Patyk and Reinhardt (1997) and include the production, as well as transport processes. They are recalculated to tonnes of nutrient and not tonnes of fertiliser. The total energy demand is calculated based on the nutrients mass flows. Different fertiliser application techniques might result in different energy demands (Tidaker, 2007). These considerations are however not included within the scope of this study.

Costs

Costs for fertilisers are not included specifically, since the benefits from selling recycled products are valued in monetary terms (see Section 4.2.6).

Please refer to Annex A.1 for a full list of the parameters used to model the Process Agriculture.

4.1.2 Households

Mass and nutrient flows

Households represent a central process for the ceMFA. The year 2007 is used as reference year, in which Hamburg had 1,741,182 inhabitants, 883,045 households and 235,623 residential buildings (Statistisches Amt für Hamburg und Schleswig-Holstein, 2007; Statistisches Amt für Hamburg und Schleswig-Holstein, 2008a). This means an average of 1.98 persons per household and 7.39 persons per building. The Process Households is used to define basic parameters relevant for further flows and processes, such as the generation of wastewater.

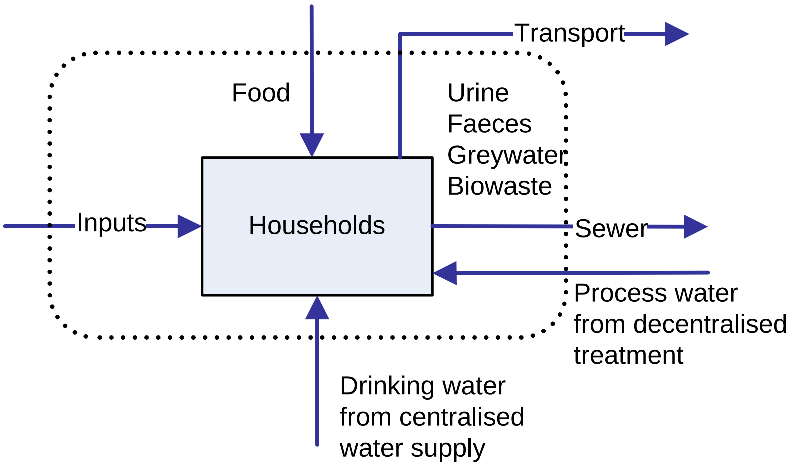

Figure 4.2: Process Households

The main inputs to households in this model are drinking water and food. Food is partly imported and partly produced in Germany. The ratio of imported to local food is based on FAO data (see Section 4.1.1 and Annex A). The water supply is modelled based on household demand, which means that for the different systems different parameters are used to determine the water consumption in the households. The water consumption varies across the systems, because, for example, toilets requiring less water for flushing are used. In general, the water supply (i.e. input) to households is calculated as the sum of water demand for the purposes listed in Table 4.2. The specific water demand values for different household activities in Germany, which are used as parameters in the ceMFA model, are summarised in Table 2.1 in Chapter 2.

Table 4.2: Purposes for water demand used in the model

Purpose	Comment
Personal hygiene, laundry, dishes, cleaning	Disposal as greywater
Gardening, car washing, etc.	Water lost in the system
Eating and drinking	Supplied by bottled water in System 5 BlaD
Toilet Flushing	Volume depending on type of toilet

In Systems 5 BlaD and 6 CompU the centralised water supply is replaced by decentralised treatment of greywater and rainwater providing process water to the households. Again, the Flow Process Water is calculated as input to the households according to demand parameters. In addition, drinking water is supplied by bottled water. For bottled water a primary energy consumption of 0.25±0.05 kWh l^{-1} is used, which includes energy requirements allocated to transport (Leist, 2002)[65]. For Systems 1

[65] Leist (2002) uses the case study of Hanover to show that the provision of bottled drinking water (112 l p^{-1} y^{-1}) consumes more energy than the supply of tap water for all purposes.

to 4, an average bottled water consumption of 128.4 l p^{-1} y^{-1} is assumed (Worldwatch Institute, 2007), whereas for System 5 an increased consumption of bottled water of 730 l p^{-1} y^{-1} is assumed due to the lack of centralised water supply.

The output flows from the Process Households comprises organic waste as well as the different fractions of the general Flow Wastewater, i.e. urine/yellowwater, faeces/brownwater, blackwater and greywater. The parameters determining these flows (mass and nutrient flows) are derived from the extensive literature review and analysis shown in Section 2.4.2. Averages and standard deviations of specific loads are used as shown in Annexes B.1 to B.5. For nutrient flows in source separated wastewater flows the daily loads per person are used as input parameter for the analysis. In addition, particularly for urine, a parameter is introduced that represents the rate of actual urine separation. Jönsson (2001) shows that this value, which normally varies between 60% and 90%, depends on the motivation of the users, as well as information and feedback. It is furthermore dependent on toilet design. The resulting urine output flow collected for recovery is calculated as follows (Equation 4-1):

$$Output_{urine} = DL_{urine} \cdot inh \cdot r_{sep} \qquad [\text{kg d}^{-1}] \qquad (4\text{-}1)$$

with
DL_{urine}: specific daily urine load [kg p^{-1} d^{-1}]
inh.: number of inhabitants [p]
r_{sep}: ratio of separately collected urine [-]

Regarding organic waste, the data quality of reported nutrient concentrations is higher than the data quality of reported daily loads. Therefore, nutrient flows from organic waste are calculated according to Equation 4-2, based on waste volume, solids content and nutrient concentrations related to solids content (see Annex B.5). Not all the organic waste is collected separately in Germany. The amount of organic waste that is separately collected is on average about 47% (Wendler, 2005). In Hamburg it is currently 22% (BSU, 2007). The ratio of separate collection is considered as parameter in order to assess the impact of higher collection rates.

$$Output_{ow} = YL_{ow} \cdot inh \cdot TS \cdot c_{nutrient} \cdot r_{sep} \qquad [\text{kg y}^{-1}] \qquad (4\text{-}2)$$

with
YL_{ow}: specific yearly organic waste load [kg p^{-1} y^{-1}]
inh.: number of inhabitants [p]
TS: total solids content of organic waste [kg kg^{-1}]

$c_{nutrient}$: nutrient concentration per TS [kg kg^{-1}]
r_{sep}: ratio of separately collected organic waste [-]

A detailed list compiling the parameters related to the process household is provided in Annex A.4.

Energy

For System 6 CompU, which includes collection of dry faeces and centralised composting of faecal matter and organic waste, the electricity demand is calculated as the sum of centralised composting with 44 kWh per tonne of input (see Section 4.1.8.1) plus the energy demand in the households for ventilation of the dry toilets. It is assumed that electric fans (8 W) are used for ventilation, resulting in a yearly electricity demand of 70 kWh per household. In the other systems no energy demand for the process households is taken into account.

Costs

The cost factors related to households, which are included in the model, are toilets and in-house piping installations. Sanitary installations such as sinks or other items that are required in every system and do not differ between the systems are not included in the analysis. Annex A.4 summarises the parameters used for the cost analysis of the process households. In Systems 3 to 6, which include source separation and/or vacuum pipes, the in-house installations are calculated based on the installation requirements given by Buchert et al. (2004). They mention pipe lengths between 0.02 m $m^{-2}_{floor\ space}$ and 0.12 m $m^{-2}_{floor\ space}$ depending on the type of urban form and the age of the building. The higher values are found particularly in rural areas so that an average value of about 0.05±0.01 m $m^{-2}_{floor\ space}$ is assumed. Using a specific floor space of 35-46 m^2 p^{-1} (calculated based on Buchert et al., 2004), this converts to about 1.8-2.3 m p^{-1}. For the calculation an average of about 2±0.4 m p^{-1} is used. The number of toilets per household varies depending on the type of building. Flats usually have only one toilet, whereas houses are often equipped with two toilets per household. Therefore, an average of 1.3±0.2 toilets per household is assumed in the calculations. For System 6 CompU energy costs are calculated based on the requirements for ventilation and are added to the operating and maintenance cost of composting toilets.

For bottled water the statistic agency of Germany cites an average cost of 0.39 € l^{-1} in 2003 (Destatis, 2006). Yet, in recent years bottled water is widely available in many shops at a lower cost of 0.13 € l^{-1} (Welt online, 2008). Therefore, this lower cost for bottled water is assumed in all systems.

4.1.3 Centralised water supply

Mass and nutrient flows

The process Centralised Water Supply is only used in the Systems 1 to 4 (CurS, NuRS, NuRu, CoDig), since in the other two systems a decentralised water supply is assumed. The mass flow going through the process is modelled in such a way that it is dependent on the water demand of the households, which varies across the systems. According to HWW (2006) the losses in the distribution system amount to about 4%, which is a bit less than the German average of 7% reported by ATT et al. (2008). Nutrient flows as well as energy demand relate to the mass flows. Nutrient concentrations in the drinking water supply used in calculations are shown in Annex A.5.

Energy

The specific electricity demand for the drinking water supply (including treatment and distribution) is set in the model to 0.51 kWh m^{-3}. This value is the specific energy consumption of the water supply system in Hamburg in the year 2007 (Hamburg Wasser, 2008b). The energy demand is comparable to other references such as, for example, Kilchmann et al. (2004), who report that the average value for electricity needed for centralised water supply in Switzerland is 0.38 kWh m^{-3} with minimum and maximum values of 0.26 kWh m^{-3} and 0.47 kWh m^{-3} respectively. In the ceMFA model the specific energy consumption is multiplied with the calculated mass flow Drinking Water.

Costs

The net cost for drinking water in Hamburg is 1.47 € m^{-3} (Hamburg Wasser, 2009). The German average, including tax and basic fees, was 1.85 € m^{-3} in 2007 (Evers, 2009). Upgrading and maintenance of assets are the main cost factors constituting about 80 to 90% of the costs (Evers, 2009). Since these cost factors are not reduced if the water consumption decreases[66], it is assumed that decreasing water consumption (as in Systems 3 and 4) has only a small impact on the total cost of water supply in systems where the centralised water supply remains. Therefore, in calculations only 20% of the water costs are set to be variable and flow-dependent, whereas the remaining 80% are fixed and are kept constant at current costs. In addition, there is a fixed charge per water meter (i.e. per household) of 23.2 € y^{-1}. If this value is related to the average number of persons per household this results in a fixed charge of 12 € p^{-1} y^{-1}. For

[66] In order to ensure proper operation and hygiene, maintenance and repair cannot be reduced if water consumption decreases. This is a problem that is apparent in many areas in Germany where decreasing water consumption may result in increased specific water costs (Herbke, 2007).

Systems 1 to 4 it is therefore assumed that the annual costs for centralised water supply in the case of existing infrastructure is given by the following Equation 4-3[67]:

$$Cost_{WS,i} = 0.8 \cdot 1.47 \, €/_{m^3} \cdot wc_{1CurS} + 0.2 \cdot 1.47 \, €/_{m^3} \cdot wc_i + 12 \, €/_{p \cdot y} \cdot inh \quad [€\ y^{-1}] \qquad (4\text{-}3)$$

with

$Cost_{WS,i}$:	costs water supply System i [€ y^{-1}]
wc:	water consumption of System 1 or System i [$m^3\ y^{-1}$]
inh:	number of inhabitants [p]

Since the cost calculation is also carried out for the case of new development, where the water supply system can be designed according to the new water demand, the costs in this case are adapted accordingly as follows (Equation 4-4):

$$Cost_{WS,i} = 1.47 \, €/_{m^3} \cdot wc_{systemi} + 12 \, €/_{p \cdot y} \cdot inh \quad [€\ y^{-1}] \qquad (4\text{-}4)$$

4.1.4 Surface areas and sewerage

Mass and nutrient flows

The Process Sewerage is only used in the Systems 1 to 4 (CurS, NuRS, NuRu, CoDig), whereas the Process Surface Areas is also part of the systems without centralised wastewater treatment.

The sewerage system of the City of Hamburg consists of mixed sewerage, i.e. domestic wastewater and rainwater is discharged in one common sewer covering an area of about 9,500 ha, and separate sewerage with separate sewers for rainwater and domestic wastewater covering 20,000 ha (Hamburger Stadtentwässerung (HSE), 2000). This means, that part of the rainwater enters the WWTP together with the domestic wastewater, whereas another part is directly discharged into surface waters or infiltrated. Excess rainwater in the mixed sewer system as a result of intense rain can cause so-called combined sewer overflow (CSO), leading to the release of polluted wastewater into surface waters. The different flows of the rainwater and sewer systems are reflected in the ceMFA model by using respective transfer coefficients for the output flows of the process Sewerage (see also Figure 4.3 and Annex A.2). The main input to

[67] It should be noted that the centralised water supply in Germany is not for profit and cost-covering, which means that all costs are reflected in the water price (bdew, 2008). Therefore, the approach of using water prices as costs for water supply is appropriate.

the sewerage is the domestic wastewater flow from the households, which is calculated based on the volume of wastewater generated in the households.

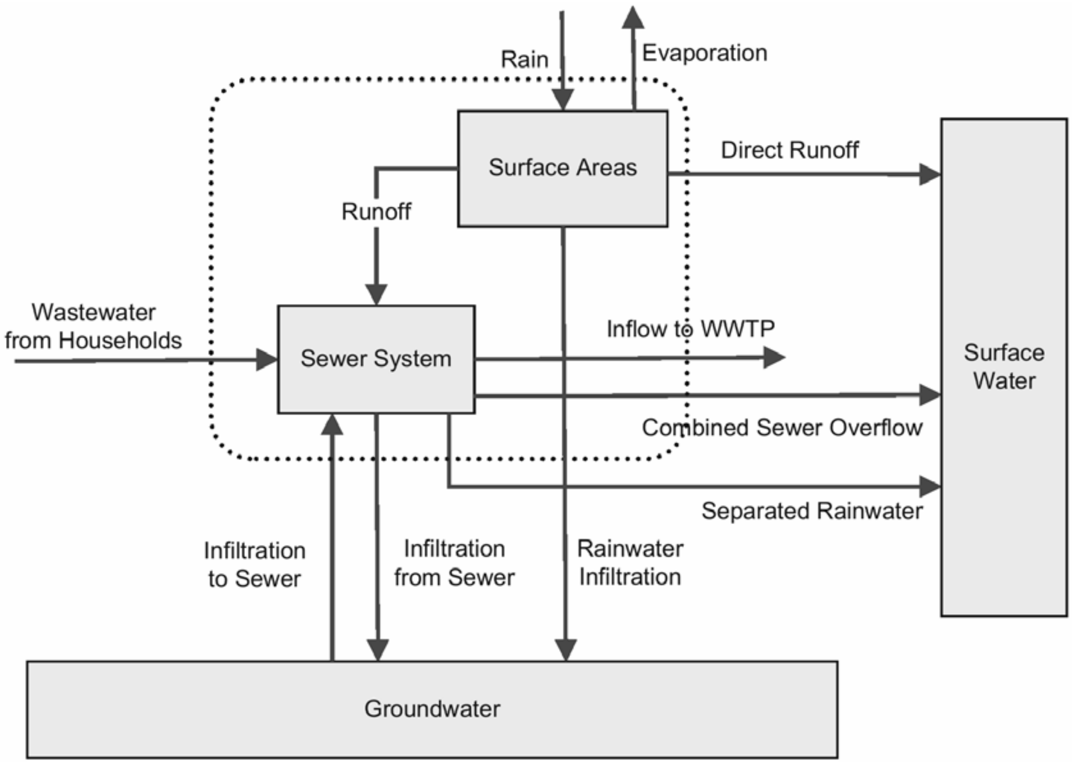

Figure 4.3: Processes Surface Areas and Sewerage

The sewer system, which for most part consists of conventional gravity sewers, together with some pressure sewers, is subjected to infiltration of groundwater to the sewer and infiltration of wastewater from the sewer into the ground. Groundwater infiltration into sewerage is difficult to determine. Values in literature vary and refer to inconsistent units, such as infiltration per sewered area, per length of sewer or per person (Franz, 2007)[68]. For this study, an infiltration rate per unit of sewered area of 700 m^3 ha^{-1} y^{-1} is calculated, based on data from Hamburger Stadtentwässerung (2000). Infiltration from the sewer into the ground is taken into account in the model as a percentage of the total flow, i.e. 2.3% of the total wastewater flow in the sewer is assumed to infiltrate into the ground (Hamburger Stadtentwässerung (HSE), 2000). Another transfer coefficient of 3%, again based on existing data (Hamburger Stadtentwässerung (HSE), 2000), is used for determining the combined sewer overflow. Since no accumulation or degradation is assumed in the sewerage, a mass balance is used to calculate the total inflow of sewerage to the WWTP. The energy demand for the process sewerage due to pumping

[68] Franz (2007) reports about average infiltration ratios of, for example, 0.1 l s^{-1} ha_{red}^{-1}, 0.1 l sec^{-1} km_{sewer}^{-1} or 55-250 l cap^{-1} d^{-1}. The calculated value used in this study of 700 m^3 ha^{-1} y^{-1} translates to 0.02 l s^{-1} ha_{red}^{-1} showing that infiltration rates in Hamburg are comparatively low.

is calculated using a black box approach. The flow entering the WWTP is multiplied by a specific electricity demand of 0.12 kWh m^{-3} (Balkema, 2003).

The Process Surface Areas, which includes neither agricultural areas nor surface waters, is introduced in the ceMFA model to allocate rainwater flows to the respective processes. Part of the rainwater evaporates, whereas another part directly infiltrates into the soil. Transfer coefficients of 0.5 (evaporation/rainfall) and 0.3 (infiltration/rainfall) are used for calculating these mass flows. The part of the runoff that enters the sewer system is calculated using a transfer coefficient of 0.216, which applies only to the sewered areas, whereas infiltration and evaporation apply to the total area. This transfer coefficient that determines the inflow to the sewerage is based on current data of rainwater in the sewer system (Hamburger Stadtentwässerung (HSE), 2000). The direct runoff to surface waters is then derived, based on a mass balance. In Systems 5 and 6 the flow Runoff to Sewerage is not included since no centralised system exists, but an additional flow Water Supply to Decentralised Treatment is introduced. This flow represents the rainwater that is used to replenish any water losses occurring in the decentralised small-scale water cycles, where greywater is treated for use as process water. In Systems 5 and 6 excess rainwater is assumed to runoff or infiltrate without any sewer system or additional treatment. However, rainwater treatment might be required in particular areas such as car parks before rainwater can be infiltrated into the soil. Yet, this process is neglected within the scope of this study.

A literature review is used for estimating the respective nutrient concentrations in the different flows. The selected values can be found in Annex A2. These concentrations are multiplied by the mass flows in order to determine total nutrient flows.

Energy

For the conventional sewerage in Systems 1 to 4[69] an electricity consumption of 0.12 kWh m^{-3} is assumed (based on Balkema, 2003). In Systems 4 CoDig and 5 BlaD a vacuum sewer system is used for the transport of blackwater. Vacuum sewers have the advantage that no infiltration into and from the sewers occurs and that also small volumes of wastewater can be transported. However, energy is required for the creation of the vacuum. Remy and Ruhland (2006) compiled the electricity demand for vacuum systems from several references. According to them the specific annual electrical energy demand varies between 7 and 51 kWh cap^{-1} y^{-1}. Considering the amount of wastewater

[69] In system 4 CoDig only greywater is discharged via the conventional sewer system, whereas blackwater is transported in vacuum sewers.

transported in the different cases this can be converted to an electricity demand[70] ranging from 3 to 28 kWh_{el} m^{-3}.The lower values are cited as being possible but such values have not been achieved in practice, whereas the higher values are a result of systems not working at full capacity. For this study an electricity demand of 15±5 kWh_{el} m^{-3} is assumed.

Costs

Costs for the Process Surface Areas and Sewerage relate to conventional sewerage in Systems 1 to 4 and vacuum sewerage in System 4 and 5. In addition, rainwater infiltration that must be taken into account in Systems 5 and 6 is covered in this section.

Investment costs for sewers are greatly influenced by various factors such as diameter, depth, soil type, material, etc. In the calculation model costs per metre for conventional sewers are approximated to average about 300±50 € m^{-1} (Guenthert and Reicherter, 2001; Oldenburg and Dlabacs, 2007). The total wastewater sewer length in Hamburg is 3,764 km (Statistikamt Nord, 2009). No differentiation is made between combined or separate sewers. Since no detailed design has been prepared, costs for pump stations are approximated by specific costs derived from a pump station design prepared by Oldenburg and Dlabacs (2007). According to their calculations pump stations require about 2 € p^{-1} for civil works (lifespan 50 years), 1.6 € p^{-1} for machinery (lifespan 12.5 years) and 0.4 € p^{-1} for electrical components (lifespan 12.5 years). A 20% error margin is assumed for these values.

Vacuum sewers (Systems 4 CoDig and 5 BlaD) have an investment cost per unit length of sewer of about 40-110 € m^{-1} (Balkema, 2003; Herbst, 2008; Oldenburg and Dlabacs, 2007). An average cost of 60±15 € m^{-1} is assumed. By approximation the length of the vacuum sewers is assumed to be about 50% of the current conventional sewers, since the buildings are connected in clusters and connecting mains can be avoided. This results in a specific vacuum sewer length of 1.1±0.2 m p^{-1}. The cost for vacuum stations are stated by Oldenburg and Dlabacs (2007) to be about 30 € p^{-1} for civil works and 30 € p^{-1} for machinery and electrical equipment. Herbst (2008) specifies the costs for vacuum stations using Equation 4-5:

$$Cost_{vacuum\_station} = 1{,}711 \cdot inh^{-0.4046} \qquad [€\ p^{-1}] \qquad (4\text{-}5)$$

with
inh: number of inhabitants connected

[70] It should be noted here that the specific electricity demand for vacuum systems for mixed wastewater is cited to be as low as 0.7 kWh m^{-3} (ATV 1995 quoted in Remy and Ruhland, 2006).

According to this formula the specific costs for vacuum stations servicing clusters of 3,000 to 5,000 persons ranges respectively between 67 and 55 € p^{-1}. Therefore, 30±3 € p^{-1} for civil works and 30±3 € p^{-1} for machinery and electrical equipment are used as cost parameters with a lifespan of 50 and 12.5 years respectively. Blackwater storage vessels are assumed to cost 125±12 € m^{-3}, of which 80% is for civil works and 20% for machinery and equipment (Oldenburg and Dlabacs, 2007).

For sewer maintenance it is assumed that the cost amount to 21 € p^{-1} y^{-1} and 19 € p^{-1} y^{-1} respectively (Wagner,2004 and Reicherter, 2003 cited in Herbst, 2008). Regarding maintenance of the vacuum sewer system another approach is followed compared to the conventional sewer system since the vacuum sewer system is smaller. For the vacuum sewer system maintenance a requirement of 1.3±0.1 € m^{-1} is assumed (Oldenburg and Dlabacs, 2007). In addition, costs for electricity used for operation of both sewer systems (as calculated in the ceMFA model) are taken into account.

In Systems 5 BlaD and 6 CompU rainwater infiltration is required in some areas, since the combined sewerage network is not in use anymore. Currently, about 9,500 ha is connected to the combined sewer. It is assumed that on 80% of these areas rainwater infiltration processes with a cost of 3±0.3 € m^{-2} and a lifespan of 20 years is applicable[71]. The remaining areas are assumed to be drained naturally or by separate rainwater pipes, which are not included in the analysis since they are the same in all systems. Operation and maintenance costs of the rainwater infiltration system are set at 3% of the investment.

4.1.5 Centralised wastewater and sludge treatment

Wastewater treatment in Hamburg is done in two combined treatment plants using the activated sludge process. The treatment includes settling as primary treatment, oxidation of organic matter and nitrogen, anoxic denitrification and chemical precipitation of phosphorus. The primary and secondary sludge is digested in anaerobic reactors, dewatered, dried and incinerated. For a more detailed description of the processes refer to Hamburger Stadtentwässerung (2000) and Albaum (1999).

[71] Halbach (2003) lists the cost range of infiltration techniques to be 2-10 € per m^2 connected surface area.

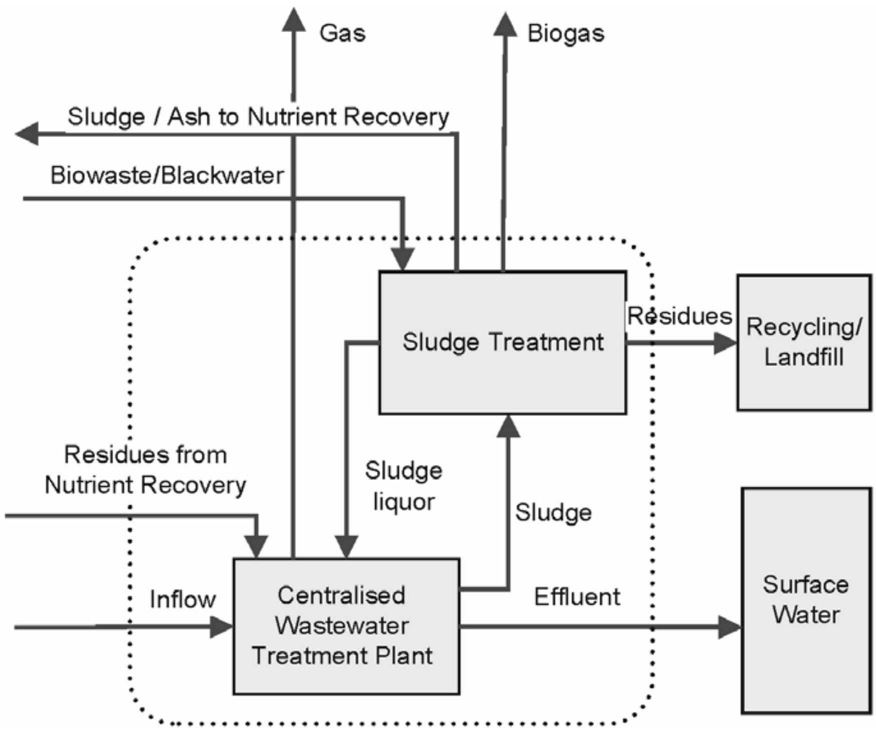

Figure 4.4: Processes Centralised Wastewater and Sludge Treatment

In this study, the mass and nutrient flows, as well as energy consumption and production, are modelled in a stationary input-output model, which is described in the next two sections. Additional data used for the calculations can be found in Annex A.6.

Mass and nutrient flows in the centralised wastewater and sludge treatment

The quantity and quality of the three input flows, namely the Flow Inflow, which is the wastewater from the households, the Flow Residues from Nutrient Recovery and the Flow Biowaste (in System 4 CoDig together with blackwater), depend on the respective system under consideration. Transfer coefficients are used to model the flows of masses and nutrients in the wastewater treatment plant as a black box model. These parameters allocate the input flows to the respective flows *effluent*, *biogas*, *gas* (air), *residues* (ash), *sludge* and *sludge liquor*. The transfer coefficients that determine the composition of the effluent with regard to carbon, nitrogen and phosphorus are adjustable parameters that are set to the current discharge standards (AbwV, 2004). Thus, these parameters (see Annex A.6) represent the removal efficiency of the WWTP, assuming that the standards can be fully met. For potassium and sulphur no standards exist. Therefore, regarding the transfer of these two substances into sludge, only the sedimentation of the particulate fraction plus the biological incorporation dependent on the eliminated organic load are considered, whereas the remaining fraction is discharged via the effluent.

In general, the calculated flow of C, N, P, K or S in the sludge is based on following relationship (Equation 4-6):

$$sludge = sludge_{primary} + sludge_{excess} \tag{4-6}$$

with:
$sludge_{primary}$: $inflow_{influent} * tc_{settling}$
$sludge_{excess}$: $(inflow_{influent+sludge\ liquor+residues\ nutrient\ rec.} - sludge_{primary} - effluent) * tc_{excess\ sludge}$
tc: : transfer coefficient for C, N, P, K or S

The transfer coefficient for settling ($tc_{settling}$) represents the particulate ratio of a substance that is removed by sedimentation in the primary settling tanks, whereas the transfer coefficient for excess sludge ($tc_{excess\ sludge}$) represents the incorporation of one of the respective substances into the activated sludge biomass. For the substances N, P, K and S, $tc_{excess\ sludge}$ depends directly on the removal of organic matter in the activated sludge reactor, since a certain ratio of the substances in the biomass is assumed. After sludge thickening, digestion and dewatering certain fractions of the substances (particularly nitrogen) are partly transferred into the sludge liquor and subsequently recycled back into the activated sludge reactor. This ratio is represented by the transfer coefficient for sludge liquor ($tc_{sludge\ liquor}$). The parameters that are used for the calculation are summarised in Annex A.6. Regarding the Flow Biogas, it is assumed that only carbon and sulphur are found in biogas, each with a transfer coefficient of 0.6±0.18 (based on Wendland, 2008). The Flow Gas represents the nitrogen that is denitrified and the carbon that is dissimilated. These flows are calculated in such a way that the maximum allowable effluent concentrations are met (see above), based on the German wastewater discharge standards, and taking into account the transfer coefficients for incorporation into biomass.

It is worthwhile noting here that according to the modelling of the Hamburg wastewater treatment plant done by Niederste-Hollenberg (2003) the activated sludge treatment process works even if the inflow is composed only of greywater resulting in limited availability of nutrients from blackwater. Therefore, the model does not need to be adjusted for the different systems that include source separation.

Energy modelling of the centralised wastewater and sludge treatment

The centralised wastewater and sludge treatment in Hamburg consists of energy-consuming, but also of energy-producing processes. Currently, 100% of the required thermal energy and about 58% of the required electricity is covered by own production (Thierbach and Hanssen, 2002).

Energy consumption

The wastewater treatment with its primary treatment, activated sludge reactors and clarifiers requires energy for processes such as pumping, aeration, mixing, etc. About 60% of the total electricity demand of Hamburg WWTP is attributed to the aeration of the activated sludge reactors for reduction of organic matter and nitrogen (Werner et al., 2007). The sludge treatment with its different processes shown in Figure 4.5 requires electricity as well as thermal energy.

The total electricity consumption of the Hamburg WWTP amounts to about 39 kWh cap^{-1} (Werner et al., 2007). For modelling the variations of energy demand in the different systems however, capita-specific values are not sufficient and a more detailed breakdown of the energy data needs to be done. For the purpose of this analysis, the energy demand of the WWTP is split up into different parameters, which are assigned to be either:

- constant (e.g. lighting, laboratory and other WWTP infrastructure)
- dependent on water flows (e.g. pumping, primary treatment, recirculation)
- dependent on the characteristics of the wastewater[72] (e.g. aeration requirements related to organic matter or nitrogen loads, precipitation of phosphorus)

The sludge treatment in Hamburg's wastewater treatment plant consists of several processes designed to maximise the energy efficiency. Thierbach and Hanssen (2002) give a description of the processes (see Figure 4.5) and analyse the overall energy balances. In order to allow the calculation of energy balances subject to varying inflows, general data obtained from literature was used[73]. In addition, data directly derived from the Hamburg treatment plant was also used. Annex A.6 summarises the energy demand parameters for the different processes. The total energy demand is calculated as the sum of the energy consumption of the different processes, which in turn are derived by multiplying the energy parameters of each process with its respective mass flows (liquid, nutrients or dry matter).

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[72] Please note that the reduction of loads due to primary treatment and overall elimination rates in the activated sludge process are considered in the model approach.

[73] For example, Müller et al. (1999) carried out an extensive study on energy utilisation on wastewater treatment plants in North Rhine-Westfalia (Germany) for the Ministry of Environment, Regional Planning and Agriculture. In addition, for some processes other more specific data is used in this study.

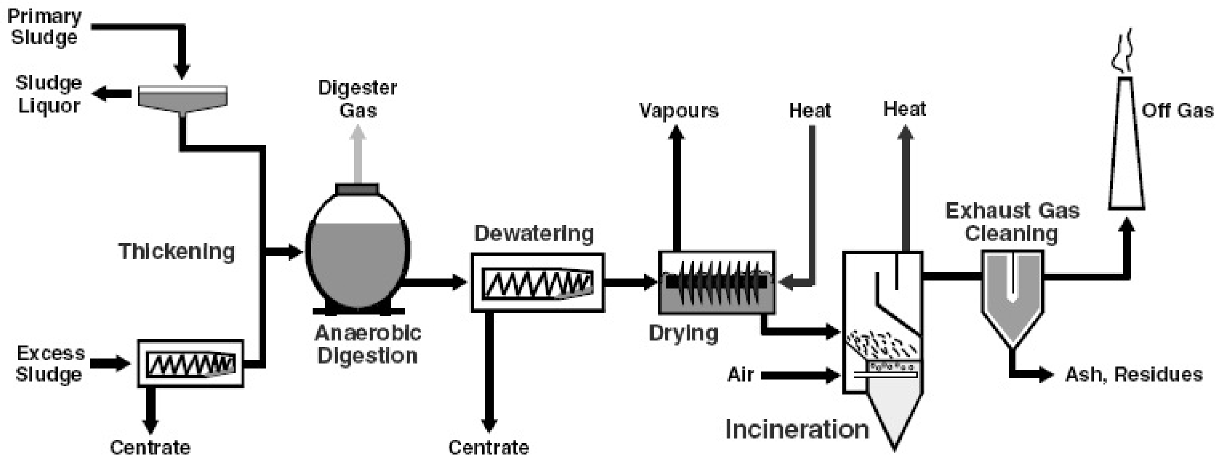

Figure 4.5: Processes of Hamburg's sludge treatment facilities (Thierbach and Hanssen, 2002)

Since the energy consumption of sludge treatment is directly dependent on the water content of the sludge, the concentration of solid matter in the sludge as it passes through the different treatment processes, plays a crucial role. The TS concentrations used in the analysis are shown in Annex A.6.

The energy demand of the anaerobic digestion process consists of heat requirements for heating up the sludge for the mesophilic process, as well as compensating for the heat losses via the exposed digester surfaces. The heat requirement for sludge heating is dependent on the mass flow, temperature difference and the specific heat capacity, and calculated as follows (Equation 4-7):

$$Q_{heat\_sludge} = m_s \cdot \Delta T \cdot c \qquad [\text{kWh y}^{-1}] \qquad (4\text{-}7)$$

with
m_s: mass flow of sludge [t y^{-1}]
ΔT: T_d – T_s, difference in temperature between reactor temperature T_d and temperature of input T_S [K]
T_d: temperature in digester, assumed to be 35°C (mesophilic)
T_S: temperature of sludge, assumed to be 12°C
c: specific heat capacity of input[74], 1.16 kWh t^{-1} K^{-1}

Heat transmission losses of the reactor depend on the externally exposed surface area, temperature difference as well as the heat transfer coefficient of the reactor and result in following heat requirements (Equation 4-8):

[74] The specific heat capacity of the input to the digester is assumed to be equal to the specific heat capacity of water (Müller et al., 1999; Wendland, 2008).

$$Q_{heat\_losses} = U \cdot \Delta T \cdot A \cdot t \qquad [\text{Wh y}^{-1}] \qquad (4\text{-}8)$$

with

U: heat transfer coefficient depending on type and thickness of surface installation, assumed to be 0.5 [W m^{-2} K^{-1}]

ΔT: T_d – T_o, difference in temperature between reactor temperature T_d and outside temperature T_o [K]

T_d: temperature in digester, assumed to be 35°C (mesophilic)

T_S: outside temperature, assumed to average at about 10°C

A: surface area of digester[75] [m^2], assumed to be 10 * 2000 m^2

t: time [8760 h y^{-1}]

Electricity required for mixing of the biogas plant is about 4±0.5 W $m^{-3}_{reactor}$ (Leschber and Loll, 1996). For the Hamburg WWTP energy requirements for mixing total about 0.32±0.04 GWh_{el} y^{-1}.

Energy producing processes

The anaerobic digestion of sewage sludge (in Systems 3 NuRU and 4 CoDig together with the additional Flows Organic Waste and Blackwater) results in the production of biogas. Biogas has a lower heating value of about 6.3±0.3 kWh m^{-3} depending on the methane concentration (Thomé-Kozmiensky, 1995; Köttner, 2005). The volume of produced biogas depends directly on the content of organic matter in the substrates and can be assumed to be the same for sewage sludge, organic waste and blackwater, namely about 0.5±0.05 m^3 kg^{-1}_{VS} (Müller et al., 1999; MURL, 1999). The organic matter contents assumed in this analysis are shown in Annex A.6 as volatile solids (VS). The energy produced by anaerobic digestion is calculated according to following Equation (4-9):

$$Q_{biogas} = \sum (m_i \cdot VS_i \cdot G \cdot LHV_{gas}) \qquad [\text{kWh y}^{-1}] \qquad (4\text{-}9)$$

with

m_i: mass flow of substrate i [kg y^{-1}] (i= sewage sludge, organic waste or blackwater)

VS_i: ratio of organic matter content in substrate i [kg_{VS} kg^{-1}]

G: gas production [m^3_{biogas} kg^{-1}_{VS}]

LHV_{gas}: lower heating value biogas [kWh m^{-3}]

[75] The WWTP in Hamburg has 10 egg-shaped digesters with a volume of 8,000 m^3 each.

On the one hand the mono-incineration of sludge requires fuel for the combustion of the sludge (see above), while on the other hand thermal energy is produced. The lower heating value of sewage sludge with a solids content of about 42% is approximately 1.2±0.1 kWh kg^{-1} (Thierbach and Hanssen, 2002; BUWAL, 1991). The energy produced can be calculated from Equation 4-10:

$$Q_{incineration} = m_s \cdot LHV_{sl} \qquad [\text{kWh } y^{-1}] \qquad (4\text{-}10)$$

with
m_s: mass flow of sludge [kg y^{-1}]
LHV_{sl}: lower heating value sludge (at TS=42%) [kWh kg^{-1}]

The produced energy (e.g. biogas and steam from incineration) is used on-site for covering the energy demand of some of the sludge treatment processes. A combined gas and steam turbine process, as well as heat exchangers are in place to produce electricity and heat. The electrical efficiency of the overall process amounts to about 20-22%, and the thermal efficiency ranges from 58-66% (Thierbach and Hanssen, 2002). Therefore, an overall efficiency of the produced energy of 78-88% is assumed in the energy model.

Costs

Costs for wastewater treatment in centralised plants vary according to factors such as the size of the plant, utilised capacity, etc. Even for the same type and size of wastewater treatment plant, specific costs can differ up to 150-200% of the average value (Reicherter, 2001). The specific operation costs decrease significantly with increasing plant size. For example, the average specific cost for a plant categorised as size 2[(76)] is about 41 € cap^{-1}, whereas for a plant categorised as size 5 it is 14 € cap^{-1} (Reicherter, 2001).

In order to be able to model the impact of reduced nutrient inflows on operation costs, an approach is required that mirrors the costs associated with different treatment processes, such as nitrogen removal or phosphorus removal. Dockhorn (2007) attributed the costs occurring at a wastewater treatment plant (including sludge treatment) to the cost units *flow rate*, *COD removal*, *nitrogen removal* and *phosphorus removal*, resulting in the cost allocations shown in Table 4.3. These ratios were applied to the total operation costs of the Hamburg wastewater plant in 2008, i.e. 41.8*10^6 € (Hamburg Wasser, 2008a). The respective operation costs have subsequently been

[76] Category 2 includes plants with COD equivalents of 1,000 to 5,000 persons, and category 5 includes COD equivalents of more than 100,000.

related to operation data, such as flow rate, or rate of COD, or nutrient removal in order to arrive at specific unit costs. These are used in the ceMFA in combination with the material flow data. A comparison with unit costs cited in Grünebaum et al. (2006) (see Table 4.3) shows a good relative correlation, keeping in mind that the latter unit costs also include capital costs. In Systems 3 NuRU and 4 CoDig organic waste is added to the digestion process. According to Arlt (2003), costs for co-digestion of organic waste amount to about 166±20 € t^{-1}_{TS}.

Table 4.3: Unit costs for COD, N and P removal in the wastewater treatment plant

		Cost unit			
	Total	Q	COD	N	P
Breakdown of operation cost [a] [%]	100	35.9	45.4	11.3	7.4
Operation cost (2008) [10^6 € y^{-1}]	41.8[b]	*15.01*	*18.98*	*4.72*	*3.09*
Operation data [b] [10^6 m^3 y^{-1}] or [10^6 $kg_{removed}$ y^{-1}]		164.63	125.87	4.07	1.31
Specific product cost [€ m^{-3}] or [€ $kg^{-1}_{removed}$]		*0.09*	*0.15*	*1.16*	*2.36*
Product costs including depreciation [c] [€ $kg^{-1}_{removed}$] (for comparison only)			0.4 (0.1-1)	5 (2-30)	10 (3-40)

[a] Dockhorn, 2007; [b] Hamburg Wasser, 2008a; [c] Grünebaum et al., 2006
(italic values represent calculated values)

In addition to operation costs, yearly re-investment costs or upgrading are taken into consideration. However, for re-investment costs no cost allocation related to pollutant loads is done, but it is assumed that these re-investments are the same for all systems with a centralised treatment. According to BSU (2009), Hamburg Wasser invested about $19*10^6$ € yearly over the period 2006 to 2008 into centralised wastewater and sludge treatment.

Analogue to operation costs, investment costs per population equivalents also decrease with increasing size of the treatment plant. In general, the specific total investment costs of newly constructed wastewater treatment plants with more than 10,000 population equivalents amounts to about 250±80 € $p^{-1}_{equivalent}$ in the year 2000 (Reicherter, 2001). Using a building-cost index of 115.3 (2000 to 2008) (Blum, 2009), this can be converted to 288±92€ $p^{-1}_{equivalent}$. Sixty percent of these costs are assumed to be for civil works with a lifespan of 50 years and 40% are assumed to be for equipment and machinery with a lifespan of 12.5 years. The separation of urine or blackwater results in reduced requirements for treatment plant construction. For example, the activated sludge tank can be designed to be smaller if shorter sludge ages need to be achieved due to reduced nitrogen elimination requirements. Peters (2007) showed for a case study in Berlin that full implementation of urine separation could reduce the activated sludge tank volume by 67%. Dockhorn (2007) also carried out a study looking at the impact of separation of urine or blackwater on the design of a centralised wastewater treatment plant. He concluded for his case study that full urine separation would require only about 41% of the investment costs as required for a conventional treatment plant. Separation of blackwater would result in 25% of the costs of a conventional plant. Therefore, the specific investment costs for the treatment plant in System 3 NuRU are set to:

$$0.41 \cdot 288 \pm 92 \; €/p_{equivalent} = 118 \pm 38 \; €/p_{connected} \quad ^{(77)} \tag{4-11}$$

For System 4 CoDig, however, it needs to be considered that blackwater is still treated in the sludge treatment. Therefore, it is assumed that the total costs are reduced to 30%. The specific investment costs in System 4 amount to:

$$0.30 \cdot 288 \pm 92 \; €/p_{equivalent} = 86 \pm 28 \; €/p_{connected} \tag{4-12}$$

Re-investments for these two systems are reduced to 7.8 mio € y^{-1} and 5.7 mio € y^{-1} respectively.

4.1.6 Decentralised treatment

In the two systems 5 BlaD and 6 CompU there is no centralised treatment of wastewater in place. In System 5, blackwater together with organic waste is treated anaerobically (see Section 4.1.6.1) and the digested slurry is transported to agriculture. In System 6, urine is separated, and faeces and organic waste are composted (see Section 4.1.8.1). In both systems, greywater is treated in a decentralised way and recycled back to the households as process water (see Section 4.1.6.2.)

4.1.6.1 Anaerobic digestion of blackwater (5 BlaD)

Blackwater, which is transported via vacuum sewers, plus organic waste, which is separately collected and shredded, are added to anaerobic digesters. These digesters serve a cluster of households with 500-700 inhabitants. Pasteurisation is achieved

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[77] Please note that due to the separation of wastewater flows the term "population equivalent" is not applicable any more. Population equivalent usually refers to the respective COD load, however, source separation results in a decrease of this load. Therefore, the actual number of people connected to the treatment plant are used as reference parameter.

through heating of the substrate to 55°C[78]. The digestion takes place in a mesophilic reactor (37°C) with a retention time of 20 days. The biogas is utilised in a combined heat and power unit (CHP). The digested effluent (slurry) is stored and used as fertiliser in agriculture.

Mass flows

Data for the characteristics of the inflow Blackwater has been derived from an extensive literature study (see section 2 and Annex B). The volume of blackwater is calculated as the sum of urine, faeces and flush water used for vacuum toilets. The organic waste load (see also Annex B.5) includes a parameter for collection efficiency, since not all the organic waste might be added to the digesters, but some might be discharged to home composting or to mixed waste disposal. Based on Effenberger et al. (2006), changes in mass flow after digestion are neglected. Also nitrogen, phosphorus and potassium loads are not affected by digestion (Sasse, 1998). Sulphur is converted into H_2S and transferred to biogas with an assumed transfer coefficient of 0.8±0.14 (calculated from data by Wendland, 2008). Total organic carbon (TOC) is converted to biogas (mainly CH_4 and CO_2) with a transfer coefficient of 0.57±0.05 (Wendland, 2008). Carbon dioxide is represented in the model as a "loss" of TOC by degradation in the digester, whereas methane is transferred to the biogas flow. The ratio of methane in the biogas is assumed to be in the range of 0.65±0.10 (Backes and Reichmann, 2000; Peters, 2002; Wendland, 2008).

Energy

On the one hand the anaerobic digester consumes energy to heat up the substrate and to compensate for heat losses through the reactor surface; on the other hand energy is produced in the form of biogas. Several authors (Peters, 2002; Wendland, 2008) conclude that heating up the substrate to pasteurisation temperatures of more than 55°C is sufficient to compensate any heat losses in small-scale digestion processes; these losses are therefore neglected in calculations. The energy required for heating up the inflows to the required temperature is calculated based on Equation 4-13:

$$Q_{heat} = m_{bl} \cdot \Delta T \cdot c_{bl} + m_{ow} \cdot \Delta T \cdot c_{ow} \qquad [\text{kWh y}^{-1}] \qquad (4\text{-}13)$$

with
m_{bl}: mass flow of blackwater [t y^{-1}]
m_{ow}: mass flow of organic waste [t y^{-1}]

[78] According to BMU (1998) pasteurisation at 55°C for 24 hours or at 70°C for one hour is required.

ΔT: $T_d - T_{in}$, difference in temperature between reactor temperature T_d and temperature of input T_{in} [K]

T_d: temperature in digester, assumed to be 55°C

T_{in}: temperature of organic waste and blackwater, assumed to be 15°C

c_{bl}: specific heat capacity of blackwater: 1.16 kWh t^{-1} K^{-1} (assumed to be the same as of water)

c_{ow}: specific heat capacity of organic waste: 1.16 kWh t^{-1} K^{-1} (dependent on water content of substrate, here assumed to be the same as water)

The shredding of organic waste added to the digester is assumed to consume 1.2±0.05 kWh t^{-1} (Vogt et al 2002 cited in Peters, 2002). Electricity consumption for stirring and pumping are estimated to be about 46±2Wh p^{-1} d^{-1} (Schneidmadl (1999) cited in Peters, 2002). Parameters used for the calculation of the energy production in the anaerobic digesters are shown in Annex A.11.

Costs

Costs for anaerobic digestion of blackwater and organic waste are estimated from calculations done by Oldenburg and Dlabacs (2007). Annex A.11 summarises the respective parameters. In addition, any costs or benefits related to energy requirements or production (heat and electricity), are included in the analysis based on the energy balances. If the generation of biogas exceeds the energy requirements of the digestion process, then a financial benefit from energy sales can be gained. Conversely, if there is an energy deficit, energy needs to be purchased. Costs for thermal energy are estimated at 0.04±0.01 € kWh^{-1}. Potential benefits from excess biogas production are reimbursed[79] at 0.08±0.01 € kWh^{-1}.

4.1.6.2 Recycling of greywater (5 BlaD & 6 CompU)

In systems 5 and 6 decentralised processes consisting of membrane bioreactors (i.e. ultrafiltration plus suspended growth bioreactor) and disinfection by UV are included, that allow the recycling of greywater as process water to the households. These systems are very compact and can be installed in individual buildings or as common plants. Rainwater is used to replenish any water losses.

Mass and nutrient flows

The input to the recycling plant is on the one hand the greywater produced in the households, and on the other hand, rainwater used to replenish any water losses (i.e. water consumed for drinking, watering flowers, etc.) and flush water in the case of

[79] Based on Renewable Energy Law (BMU, 2009).

System 5 (vacuum toilets for BlaD). Research is currently underway with regard to the treatment efficiency of membranes for greywater treatment and process water recycling (Gnirss et al., 2003; Li et al., 2008). Since the modelling of the membrane treatment is not within the scope of this study, it is assumed that the effluent used as process water in the households has the same characteristics with regard to C, N, P, K and S, as the drinking water supplied by the centralised water supply in the other systems. Organic matter and nitrogen are removed by biological processes, whereas phosphorus is partly incorporated into the biomass and partly precipitated.

Energy

Energy demand in membrane bioreactors (for aeration, as well as for the pressure in the membrane) varies depends firstly on the kind of membrane used (i.e. pore size) and secondly on the type of wastewater treated. The electricity consumption for membrane bioreactors stated by several authors varies between 0.4 and 1.5 kWh m^{-3} (Kionka, 2008; Nolde et al., 2007; Peter-Fröhlich et al., 2007; ATV-DVWK, 2005). UV disinfection is stated to consume about 0.1 kWh_{el} m^{-3} (Altepost, 2003). In addition, there is energy needed for pumping the process water to its destination. In the model, the total electricity demand for greywater treatment and reuse is assumed to be 1.5±0.2 kWh m^{-3} in System 5 BlaD, and 2±0.2 kWh m^{-3} in System 6 CompU, where higher treatment levels are achieved.

Costs

According to literature and manufacturers information, costs of membrane bioreactors for greywater recycling vary widely between 200 and 1800 € p^{-1}, depending mainly on the size of the plant (see, for example, fbr, 2005; Kaufmann et al., 2007; Starkl et al., 2005; Kionka, 2008). Yet overall the cost trend is currently strongly decreasing, since membranes are increasingly used in water and wastewater treatment. Based on this trend a value of 300±50 € p^{-1} for System 5 BlaD and 400±50 € p^{-1} for System 6 CompU, where UV disinfection is included, is assumed. Structural parts (50% of the plant) are assumed to have a lifespan of 20 years, while machinery and equipment is assumed to have a lifespan of 12.5 years.

Operation costs for greywater recycling include the replacement of membranes, energy, chemicals for cleaning the membranes and maintenance. In the cost calculation model, costs for electricity are based on the energy consumption. In addition, operation and maintenance costs are assumed to be 2.5% of the investment costs. UV lamp servicing is assumed to cost about 6±0.5 € p^{-1} y^{-1} (based on Boller, 2006).

4.1.7 Transport

Source separation of wastewater flows has the advantage of smaller volumes, which are easier to handle. However, the increased number of flows requires alternative means of transport instead of sewer systems. Thus, the importance of lorry-based transport for the analysis of source-separating wastewater systems is obvious. Other studies on transport in wastewater system analysis usually used a fixed distance between point of origin (i.e. household) and destination. Tidaker et al. (2007) and Wilsenach et al. (2003) both assumed a distance of 10 km between households and treatment facilities, or farms for the case of Sweden and the Netherlands respectively. Wittgren et al. (2003) and Maurer et al. (2003) calculated the environmental impact with a transport distance of 50 km and 60 km respectively. Jönsson (2002) calculated the break-even distance, which is the distance that urine could be transported without undoing the energy gains in the wastewater treatment plan, to be 95km, whereas Bengtsson et al. (1997, cited in Jönsson, 2002) calculated it to be 60km.

However, regular collection of urine or blackwater from households, which is similar to the collection of domestic waste, is a logistical challenge requiring a more detailed investigation of transport needs. Therefore, the analysis of transport requirements goes beyond the simplified assumption of travel between point A and point B. In the following sections the model for transport needs assessment used in this study is presented.

4.1.7.1 Lorry-based transport in the system analysis

Transport requirements are required in the different systems for different flows, i.e. urine, faeces, blackwater and organic waste, as well as for different trips (Figure 4.6). Since volumes of the different flows (i.e, urine, blackwater and organic waste) vary, the logistical aspects differ and therefore transport requirements need to be calculated separately for the respective flows. In order to simplify the analysis it is estimated that generally the distance to centralised facilities like the wastewater treatment plant and the composting plant is the same (see trip 4 and trip 1). Storage and nutrient recovery facilities are assumed to occur in four "stations" distributed throughout the city (see Section 4.1.7.2). Trips 1 and 3 refer to transport from households to several stations, whereas trips 2, 4 and 5 are from households to a single central plant. Concerning the transport of flows from central facilities to agriculture ("delivery"), a fixed distance of 25 km[80] one way is assumed (trips 6 to 10).

[80] This distance is based on following assumptions: Agriculture as a ring around Hamburg, starting at a distance of 20km (i.e. circle radius). In total about 5000km² of agricultural land is required. This results

For the detailed calculation and analysis of the collection of source separated wastewater flows from households or neighbourhood facilities (diffuse sources, trips 1 to 5), a GIS-based model is developed using the software ArcGIS. The GIS model represents a simplified method for the assessment of logistics involved in the lorry-based collection of different source separated flows. For operational purposes and least-cost modelling, an approach focussing more on trip planning and route optimisation[81] would need to be used. For the purposes of this study however, an assessment of overall travel routes is considered to be sufficient.

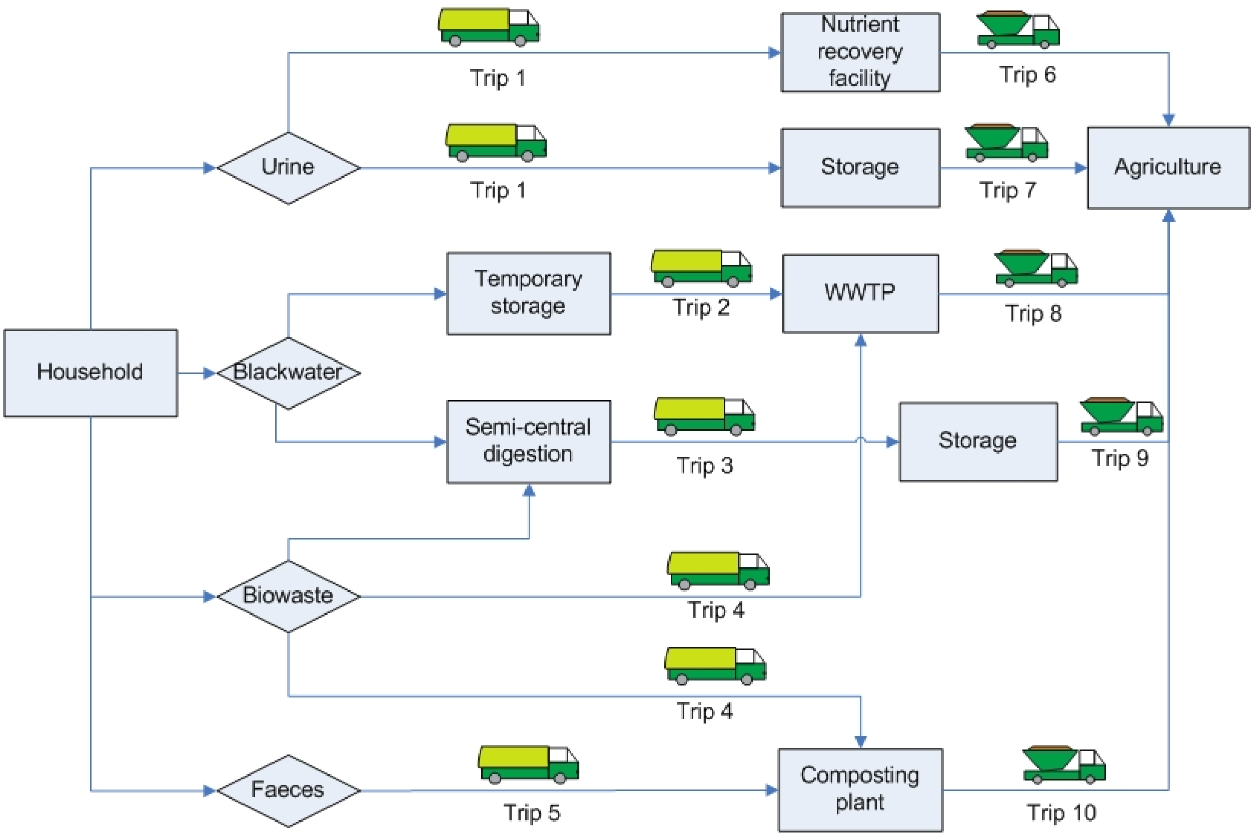

Figure 4.6: Summary of transport included in the systems[82]

The further use in agricultural areas can be either coordinated by centralised distribution centres or by individual farmers. Transport requirements can therefore vary according to the subsequent distribution and final application on agricultural fields; this is however not included in the analysis.

in an "outer ring" with a radius of about 45 km. Therefore, a distance from the periphery of Hamburg of 25 km is assumed as an approximation.

[81] See for example the "travelling salesman problem" as used in operations research methods (Bousonville, 2002).

[82] Trips with the same number are assumed to have the same distance.

4.1.7.2 Data and assumptions used for the transport model

Hamburg is administratively structured into seven wards each with several districts, which in turn are divided into different neighbourhoods. Detailed data on these administrative units (e.g. size, inhabitants, housing types, etc.) is available from the statistics agency (Statistisches Amt für Hamburg und Schleswig-Holstein, 2007). For this analysis the year 2006 is used as reference year. A digital layer map is available showing the street system of Hamburg including the length of the individual streets. In total, Hamburg's street system has a length of about 4640 km. A second digital layer including the 103 districts of Hamburg is used to allocate the streets to the respective districts in the GIS. The street data is not prepared as a network but as separated lines which cannot be combined to a network due to the complexity. Thus, it is not possible to use the network analysis within ArcGIS, but a combination of analyses in ArcGIS and MapPoint is used.

The source separated wastewater is collected from the households or from neighbourhood facilities, like storage vessels or cluster treatment plants. It is then transported to further treatment or storage (see Figure 4.6). These secondary stations are assumed to be depots for the collection lorries so that the lorry trips start and end at the respective stations. In Systems 2 NuRS and 4 CoDig the existing central WWTP is assumed to be the only station (see trip 2 in Figure 4.6). The location of the existing plant south of the river Elbe however, is restricting the access for lorries, since this would lead to enormous congestion problems concerning the access roads to the plant, if traffic increases due to transport of urine or blackwater. Therefore, it is assumed for Systems 3 NuRU, 5 BlaD and 6 CompU that four such stations exist across the city. This is in line with the four currently existing maintenance and storage facilities of Hamburg Wasser, which could possibly be converted into treatment facilities for blackwater or urine[83]. For the analysis, the different districts are allocated to one of the four stations (Bergedorf, Harburg, West or North). This is done using ArcGIS analyses based on least distance. This results in the allocation of districts and inhabitants as shown in Annex A7.

The transport needs are generally composed of trips going to the area where the respective flow (i.e, urine, blackwater, organic waste) is collected and trips within the area until the vehicle is fully loaded. These are called in the following sections "access trips" and "collection routes" respectively. The total transport distance is therefore the sum of the access trip from the depot to the neighbourhood (and return) and the

[83] Also the Hamburg waste management department has five depots within the town borders.

collection route within the neighbourhood. Figure 4.7 illustrates the calculation algorithm using ArcGIS, MapPoint and Excel commands.

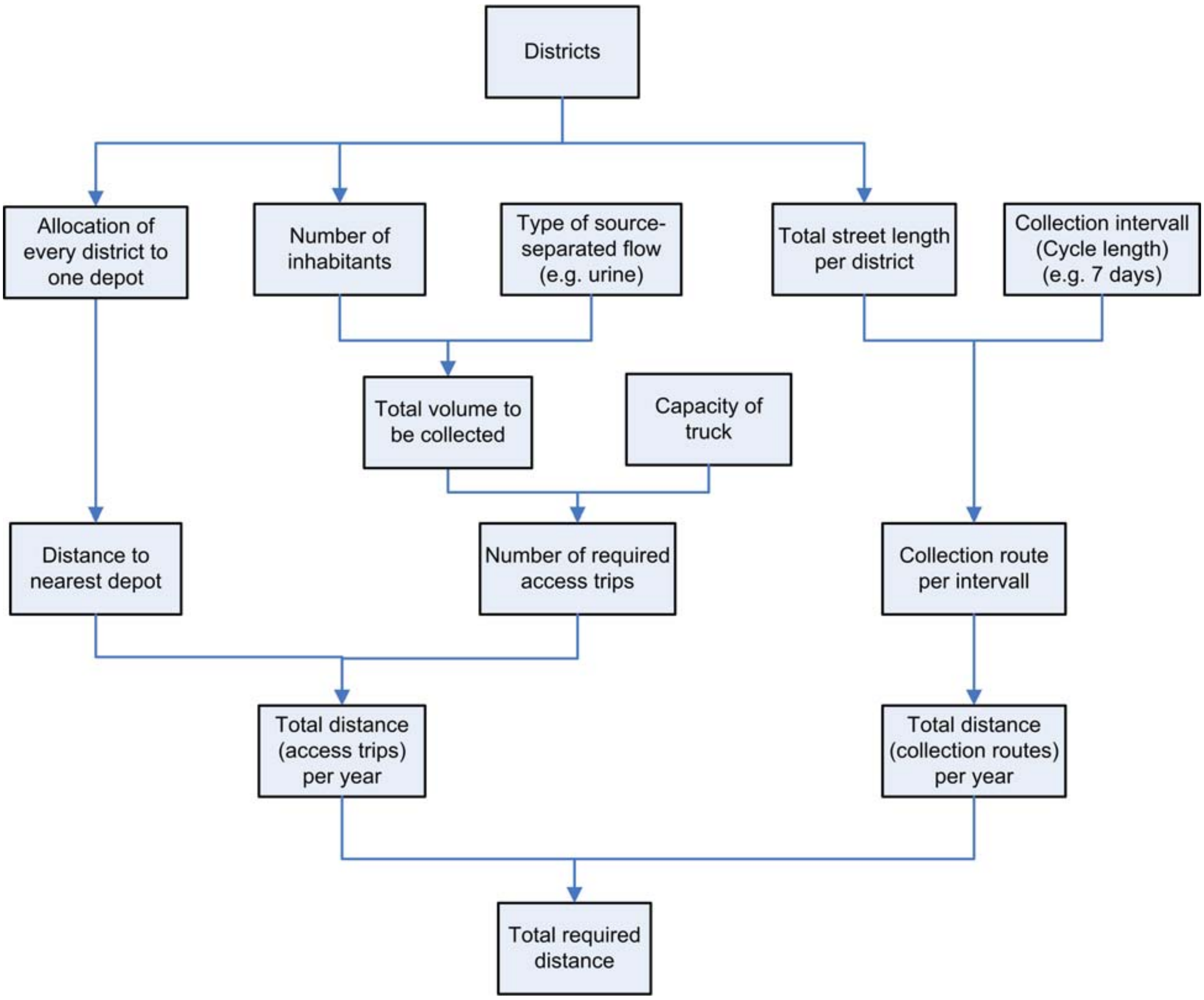

Figure 4.7: Calculation procedure for transport requirements from households to processing stations

Access trips

The distance between the households/neighbourhoods and the secondary storage and processing facilities (access trips) is analysed using the route planning possibilities of the software MapPoint. First, the geometric centre of every district is determined using a layer analysis in GIS. Subsequently, the distance and travel time spent between the centre of each district and the central WWTP, as well as the four stations, is calculated in MapPoint, based on the street system of Hamburg.

The required access trips and their total length in one collection cycle are calculated by following algorithm:

$$N_{aj} = inh_j \cdot vol \cdot d \cdot M^{-1} \qquad [-] \qquad (4\text{-}14)$$

with
N_{aj}: number of access trips to district j [-]
inh_j: number of inhabitants in district j [p]
vol: volume or load to be collected (per person and day) [kg p^{-1} d^{-1}] or [m^3 p^{-1} d^{-1}]
d: collection intervall (i.e. every d days) [d]
M: maximum load of collection lorry [kg] or [m^3]

$$L = \sum N_{aj} \cdot D_{Dist} \qquad [km] \qquad (4\text{-}15)$$

with
L: total length of access trips in one collection cycle [km]
N_{aj}: number of access trips to district j [-]
D_{Dist}: distance between district/neighbourhood and depot [km]

The maximum load M of a collection lorry is set to be 11 tonnes for solids collection and 16 m^3 for liquids collection (based on Giese, Th., Hamburg Wasser, personal communication, 20 Nov 2006 and Hamburg Sanitation Department cited in Grünauer, 2007). A fortnightly collection interval d is assumed.

Collection routes

For the calculation of the collection routes it is assumed that every road needs to be fully serviced by the lorries and, thus, the full length of all roads is included in one collection cycle. On the one hand this might result in some overestimations, since source separated flows from houses in smaller roads might be drained in pipes towards the collection containers on main roads. This means that lorries would not need to drive along these small roads to empty the containers. On the other hand the collection containers might be located on the left and on the right side of large roads requiring a bidirectional collection trip, which is also not included in this assumption. Therefore, taking into account that every road will be serviced once represents a trade-off between these considerations.

Results and discussion of the transport model

The distance travelled resulting from the transport model is shown in Annex A7. As expected, the results show that the transport of liquids such as urine or blackwater is considerably greater than other transports. Particularly blackwater, due to its relatively high volume, requires extensive transports. Comparing trips number 2 and 3 emphasises the need to consider the implementation of several distributed storage and treatment centres instead of one centralised facility. For the case of Hamburg, four semi-centralised facilities decrease the distance travelled per year by about 56% as compared to a single facility. It should be noted that the distance travelled shown here is of course dependent on underlying assumptions such as a fortnightly collection cycle, the amount of flushwater used, separation effectiveness, etc.

Another point of interest is to check whether the accuracy increases significantly when using this detailed calculation procedure, instead of using average transport distances. The MapPoint analysis shows that for the case of Hamburg the average distance from every neighbourhood to the nearest of the four treatment facilities is about 9.2 km, whereas the average distance to the centralised treatment plant is about 17.8 km. Using the result for the urine mass flow per year (565,000 m^3 y^{-1}, see Annex C), a lorry capacity of 16 m^3 and an average distance (one-way) of 9.2 km, this would result in a total distance of 649,750 km y^{-1} as compared to 693,879 km y^{-1} as derived from the detailed analysis(see Annex A.7). The comparison of required distance travelled for blackwater transport to a centralised facility shows a similar picture (10,502,000 km y^{-1} compared to 11,084,050 km y^{-1}). This means that for both cases the relative difference obtained by using the simplified calculation is about 5% to 6%. It is therefore suggested that for future analyses the differentiation between access trips and collection routes as well as the detailed GIS-based model can be neglected and that average distances are sufficient.

Energy

Energy requirements are calculated based on the required travel distance multiplied by the fuel consumption and its respective energetic value. Regarding fuel consumption, it is taken into account that the lorries are empty while going forth and full on return. The data that is included in the energy model is summarised in Annex A.7.

Costs

Costs for transport could be calculated in a detailed manner using factors such as transport distances, average speed, time for loading and unloading, labour costs etc. To simplify the calculation an average total cost per km is used in the cost model. Arlt (2003) compared standard prices and own calculations regarding the logistic costs of waste and sewage sludge transport. From his results it can be concluded that average costs for lorry trips between 25 and 50 km are about 200±20 € $trip^{-1}$ and for trips between 75 and 100 km are about 300±20 € $trip^{-1}$. This can be converted to costs between 3 and 8 € km^{-1}, depending on the trip length. For the calculation an average of 4±0.5 € km^{-1} is assumed; this includes capital costs, consumables, labour and maintenance.

4.1.8 Nutrient recovery processes

4.1.8.1 Composting (1 CurS, 2 NuRS, 6 CompU)

The process Centralised Composting is used in systems 1 CurS, 2 NuRS and 6 CompU, producing the outflow compost which is used in agriculture for the provision of organic matter, and to a lesser extent also nutrients. SImilar to the current compost plant in Bützberg where part of the organic waste of Hamburg is composted, a centralised composting plant is assumed. In System 6 CompU the aerobic degradation of organic waste is extended to also include faecal matter.

Mass and nutrient flows

The nutrient and mass flows are modelled using transfer coefficients as they are depicted in Figure 4.8. The transfer coefficients are applied since the final product compost contains only parts of the nutrient and organic matter of the input. The remainder is lost either to the gas phase or as leachate, which are both integrated in the ceMFA as Flow Residues. In general, there is a rather high variation of the coefficients, which is reflected in the error margin for the transfer coefficients. For example, nitrogen emissions are dependent on the C/N ratio of the input, as well as the temperature (Amlinger and Götz, 2000). With a lower C/N ratio higher ammonia emissions can be expected (Gotaas 1956 cited in Jenkins, 2005). For a detailed list of the transfer coefficients and their sources, please refer to Annex A.3.

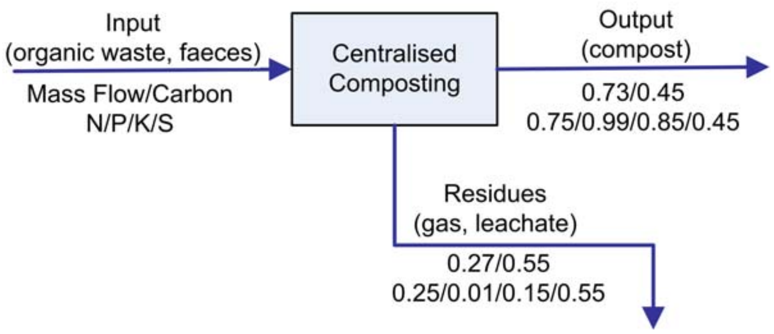

Figure 4.8: Transfer coefficients for the process composting for mass flow, carbon and nutrient flows

Energy

Centralised windrow composting requires energy for different processes such as removal of impurities, turning and ventilation. Aeration is the most energy-consuming step in this process requiring about 25-40 $kWh_{el}\ t^{-1}{}_{biowaste}$. In the ceMFA model the electricity consumption is calculated using an estimated average electricity demand of

44 kWh per tonne of input based on the work of Thomé-Kozmiensky (1995) and Schmelz (2000).

In addition to centralised composting plants, home composting represents a valuable process for waste disposal as well as gardening. Since however, the volume of nutrient flows in gardening in Hamburg is not significant, home composting is disregarded in the ceMFA. However, this does not mean that it shouldn't be integrated into the planning of sustainable nutrient recycling systems and further promoted, because it represents an easy and relatively inexpensive contribution to waste management.

Costs

Costs for the composting process in large scale plants are in the range of 320±20 € $t^{-1}{}_{dry\ matter}$ (Arlt, 2003); this includes capital as well as all operation costs. In addition, a benefit from selling compost is assumed. Arlt (2003) indicates that good quality compost has a market price of 30 to 80 € $t^{-1}{}_{dry\ matter}$. This is in line with a small market survey done by Oldenburg and Dlabacs (2007) who conclude that compost has a market price of about 60 € m^{-3}. For the cost calculation a benefit of 55±5 € $t^{-1}{}_{dry\ matter}$ is assumed.

4.1.8.2 Nutrient recovery from sewage sludge (2 NuRS)

The Process Nutrient Recovery from Sewage Sludge is based on the Seaborne® system, which is currently under development (see also Section 2.4.3). The recovery processes used in the Seaborne® process are basically precipitation of magnesium ammonium phosphate (MAP) as well as subsequent stripping of ammonia by scrubbers. Digested sewage sludge from the central wastewater treatment plant is fed into the process.

Mass and nutrient flows

Magnesium oxide, sodium hydroxide and sulphuric acid are added to the processes as chemicals. The outputs of the process are recovered nutrients (products), sludge liquor (centrate) which is directed to the activated sludge tank, and residues after incineration (ash). The parameters that are used to calculate the input and output flows are summarised in Annex A.8.

Energy

The energy requirement for MAP precipitation is calculated as sum of the energy demand for magnesium production dependent on the dosed Mg per tonne P, plus the specific energy demand for the process dependent on the sludge volume. The steam stripping process requires energy for heating up the input, electricity for the stripping itself and electricity for the production of sodium hydroxide used for adjusting the pH. Detailed parameters are shown in Annex A.8.

Costs

Since the detailed design of the nutrient recovery plant goes beyond the scope of this study, the investment cost of the first pilot plant using the described set-up is used as approximation. According to Seaborne EPM AG (2009) the total investment costs including all equipment and engineering for the plant in Gifhorn, Germany was $3.7 \cdot 10^6$ €; this was for a capacity of 140 $m^3_{sewage\ sludge}$ d^{-1} or 4 t_{TS} d^{-1}. Related to the 65,000 population equivalents served by this plant, this amounts to an investment cost of about 57 € p^{-1}. Assuming that the larger plant size in Hamburg and improvements in plant design result in a decreased investment costs of about 30%, a specific investment cost of 40±12€ p^{-1} is used for this study. It is assumed that about 50% of these costs are for civil works with a lifespan of 50 years and the remaining 50% for machinery and electrical equipment with a lifespan of 12.5 years. Labour is estimated at 1% of the investment costs, and maintenance and repair at 3% of investment costs. Costs for energy, as well as materials such as NaOH, H_2SO_4 and MgO are derived based on the requirements calculated by the ceMFA model and the costs given in Annex A.8.

For selling $(NH_4)_2SO_4$ from recovery processes, a revenue of 30 € per tonne of ammonium sulphate is stated by Herbst (2008) and Dockhorn (2007). Therefore, a benefit of 58±6 € t^{-1}_N is assumed. Also, as indicated in literature, MAP can be sold for prices ranging from 100 to 250 € t^{-1}_{MAP} (Herbst, 2008; Montag, 2008; Wilsenach, 2006). Related to the recovered phosphorus load[84], a benefit of 1500±200 € t^{-1}_P is assumed.

4.1.8.3 Nutrient recovery from sludge liquor and sludge ash (4 CoDig)

System 4 CoDig includes a combination of two nutrient recovery processes at the centralised wastewater and sludge treatment. Firstly, stripping for ammonia recovery is applied to the sludge liquor. Secondly, ash from sewage sludge incineration is subjected to a process for phosphorus recovery similar to the BioCon® process (see also Section 2.4.3).

Mass and nutrient flows

The fertiliser output of the BioCon® process is about 66 kg H_3PO_4 per tonne TS (Pettersson, 2001). A detailed list of the parameters used for the mass flow and nutrient modelling of the two recovery processes, can be found in Annex A.9.

[84] It is assumed that P constitutes 12.6% of the total mass of MAP ($MgNH_4PO_4 \cdot 6H_2O$).

Energy

The energy consumption for the stripping process of the sludge liquor is based on the work of Haberkern et al. (2008), who state an electricity demand of 1.75±0.15 kWh per kg of N eliminated and a heat requirement of 9.5±1 kWh per kg of N eliminated. The required electricity for the production of sodium hydroxide is estimated to be about 1200±135 kWh per tonne of NaOH (UBA and Öko-Institut, 2000). Energy requirements for the production of H_2SO_4 and HCl are neglected, since these chemicals can be produced as by-products from other processes, or can be recycled within the processes (UBA and Öko-Institut, 2000). Other energy requirements for the BioCon® process can only be estimated by referring to other related processes, since no detailed information on electricity consumption is available in literature. The energy requirement for milling of the ash depends on the required fineness. As a rough estimate an electricity demand of 10±2 kWh t^{-1} is used in the calculation (Jorgensen, 2002; see also Lange, 2009). The power consumption of the ion exchange device is estimated to be about 0.5 kW, which results in annual electricity demand of approximately 1800±360 kWh. In addition, a generic electricity demand for the BioCon® process (for dosing, pumping, stirring etc.) of
0.5±0.05 kWh t^{-1} is assumed in the modelling.

Costs

Stark (2002) indicates that the product costs for the BioCon® process is in the range of 3.5 to 9 € kg_P^{-1}. For this study the cost parameter for the BioCon® process is set to 3±0.5 € kg^{-1}_P, assuming that process costs will level off at the lower range through research and innovation.

The cost for steam stripping is calculated by adding the costs for investment, material, labour, energy and maintenance. Capitalised investment costs are based on a specific cost of 0.72±0.1 € y^{-1} per m^3 of substrate process annually (Dockhorn, 2007). Material costs are calculated by multiplying respective mass flows and specific costs of H_2SO_4 and NaOH (see Annexes A.8 and A.9). Energy costs are made up by the thermal energy required and electricity costing 0.15 € kWh^{-1} and 0.04 € kWh^{-1} respectively. Labour costs are set to 10±1% of the capitalised investment costs. This represents a relatively high ratio, since Dockhorn (2007) states that the stripping process has high manpower requirements. According to Dockhorn (2007), maintenance and repairs are set to 20±2% of the capitalised investment costs.

The product of the BioCon® process is assumed to be sold for 1,250 € per tonne phosphoric acid (Linker, 2008). In addition, ammonia solution from the stripping process is sold for 58 € per tonne N (see Section 4.1.8.2)

4.1.8.4 Nutrient recovery from urine (3 NuRU)

The Process Nutrient Recovery from Urine is modelled as a black-box system, assuming that it consists of the two combined processes MAP precipitation and steam stripping. The input to this process is urine collected in urine diversion toilets.

Mass and nutrient flows

MAP precipitation for phosphorus and nitrogen recovery achieves high recovery rates of particularly phosphorus. Because after precipitation there is still nitrogen remaining in the substrate due to the molar ratio of N and P in MAP, steam stripping is used to recover the nitrogen. Sulphur is added to this process as sulphuric acid to absorb the ammonia and to produce the final product (=fertiliser) ammonium sulphate. The transfer coefficients used for the model represent the degree of recovery achievable by the combined process. Annex A.10 lists the parameters and transfer coefficients used for modelling the mass and substance flows of the process. After the recovery processes the remaining liquid has still a relatively high organic load and, therefore, needs further treatment. In the ceMFA model, this flow is diverted to the process *sewer*, so that it can be treated in the central wastewater treatment plant.

Energy

The energy consumption required for the combined process is calculated as the sum of energy requirements for the production of MgO, the additional energy requirements for dosing, stirring, etc. for the MAP process, and the heat and electricity for stripping. The amount of required MgO depends on the phosphorus in the input that must be precipitated. Stochiometric ratios greater than one are reported to achieve a higher efficiency. Energy requirements for stripping are needed for heating up the urine, as well as for the stripping process. Tettenborn et al. (2007) report that the main part of the energy consumed in their laboratory scale tests was for heating up, and only a 1/6 was for stripping. Therefore, with energy recovery and improved insulation, higher energy efficiencies can be achieved. The value used here as a parameter represents a theoretical value implying these considerations. The parameters used for the energy modelling are summarised in Annex A10.

Costs

Esemen and Dockhorn (2009) projected the product costs for a plant for MAP precipitation from urine using a population equivalent of 350,000. Since this plant size is about the size of one of the plants at the four stations, the cost data can be transferred to this case study. Based on the data given by Esemen and Dockhorn (2009) it can be concluded that MAP precipitation from urine costs about 530 € per tonne MAP or 4206 € per tonne P. This value includes capital costs, maintenance, labour, magnesium addition and energy, and was derived using a depreciation period of 10 years and a discount rate of 6%. An error margin of ±20% is assumed for this cost calculation.

Costs of the steam stripping plant are derived from Dockhorn (2007). He designed and costed out a steam stripping plant for 350,000 population equivalents, which resulted in a specific cost of 83 € per tonne of $(NH_4)_2SO_4$ or 955 € per tonne of $N_{eliminated}$ (exluding benefits). For this study a cost parameter of 950±50 € per tonne of $N_{eliminated}$ is applied. The benefits of selling MAP, and ammonia solution, are given in Section 4.1.8.2 as 1500±200 € t^{-1}_{P} and 58 € t^{-1}_{N} respectively.

4.1.8.5 Direct application of urine and slurry (5 BlaD and 6 CompU)

Mass and nutrient flows

In Systems 5 BlaD and 6 CompU urine, as well as slurry from digestion of blackwater and organic waste, is collected, stored and directly applied in agriculture. The mass flows are calculated, based on the effluent flows of the processes households and anaerobic digestion. Nutrient losses can occur during transport and storage and are calculated using the transfer coefficients as shown in Table 4.4. In addition, due to the high pH of stored urine, nitrogen losses can occur during agricultural application to farmlands. The corresponding transfer coefficient (i.e. loss) is set at 0.07±0.05, based on work by Benetto et al. (2009) and Kirchmann and Petterson (1995).

Table 4.4: Nutrient losses during transport and storage

	N	P	K	S
Transfer coefficient (loss)	0.01±0.002	0.20±0.02	0.01±0.003	0.01±0.003

Values based on Maurer et al. (2006a) and own assumptions

Energy

The application of flows with relatively high volumes, such as untreated urine or slurry from anaerobic digestion, requires comparatively more energy than the application of mineral fertiliser or other products with nutrients in a concentrated form. These differences are however, neglected within the scope of this analysis.

Cost

The application of urine in agriculture requires more effort in terms of labour and machinery, than the application of mineral fertiliser. Oldenburg and Dlabacs (2007) have quantified the cost of this additional effort as 33 € ha^{-1}. Roughly, an application of 10 m^3_{urine} ha^{-1} is assumed, which translates into an additional application expense of 3.3 € m^{-3}_{urine}. Effenberger et al. (2006) cite the cost for the application of slurry from a biogas digester as 1.86 € m^{-3}. Thus, in the cost calculation model for both substrates (i.e. urine and slurry), an additional cost of 2.5±0.2 € m^{-3} is assumed for agricultural application.

Since fertiliser should only be applied according to crop requirements, storage facilities for urine and slurry are required. In Germany, urine and slurry could be applied twice a year and there is therefore a need to store these fertilisers in silo installations nearby agricultural fields for six months. Investment costs for large silos are approximated by a specific cost of 100±20 € m^{-3}, with an estimated lifespan of 50 years and 1.5% of the initial investment cost for maintenance and repairs (based on Silovereinigung, 2006; dlv, 2009). Storage requirements are calculated, based on mass flows.

4.2 Results

The following sections present an overview of important results for the analyses of the six systems. The main focus of the discussion is on criteria relevant to resource efficiency, such as nutrient recovery and energy demand, but other criteria such as emissions of nutrients and organic matter to water sources are also included, since they represent criteria that are conventionally used for assessing the efficiency of wastewater systems. The uncertainties shown here refer to the variability derived from the calculations using normal distributions, i.e. which are based on Gauss' law of error propagation. In addition, Monte Carlo simulations are carried out. The results of the Monte Carlo calculation do generally not differ significantly from the results shown here, and are therefore not included in the discussion. A brief reference to the Monte Carlo simulations is made in Annex D.

4.2.1 Nutrient emissions to the environment

Nutrient emissions to the environment are included in the model as emissions to soil/groundwater, and to surface water, and as gaseous emissions in the case of nitrogen and sulphur. Groundwater contamination can result from infiltration of wastewater from sewers into the ground, infiltration of rainwater runoff from surface areas and infiltration from agricultural areas. Nutrient flows to surface water are from various sources, namely rainwater from separate sewer systems, wastewater from combined sewer overflow, effluent of the wastewater treatment plant[85], rainwater runoff from surface areas, erosion and runoff from agricultural areas. Gaseous emissions can occur

[85] According to UBA (2003) nitrogen emissions from German wastewater treatment plants and sewers contribute 17% and 4% respectively of the total emissions into surface waters. Regarding phosphorus these emissions are about 35% and 10% respectively.

at the wastewater treatment plant and the composting facilities. Emissions into air are however, beyond the scope of this study and are only briefly discussed here.

Emissions can be used as an indication of potential environmental pollution. However, they can only give an indication of the risk. For example, infiltration of nitrogen into the soil may result in serious groundwater pollution. If however, the distance to the aquifer is sufficient and intervening soil properties are favourable, this risk might be minimal. Although considerations relating to risk assessment have not been included in this study, total loads are used as indication.

Nitrogen

Figure 4.9 illustrates the total nitrogen emissions to groundwater and surface water from the water-system related processes of the six investigated systems. The total emissions to groundwater and to surface water are summarised in Table 4.5. The effluent of the wastewater treatment plant, although adhering to current standards, contributes the largest share of emissions (about 84-86% of the total discharges to surface waters)[86]. These emissions can be reduced by 17% (3 NuRU) and 42% (4 CoDig) through the introduction of source separation of urine and blackwater, respectively. Source separation can in addition reduce pollution from infiltration of wastewater from sewers into the ground, as well as from the combined sewer overflow. If, for example, blackwater is transported in separate vacuum sewers (4 CoDig), nitrogen emissions from both sources can be reduced by 90%. The reduction of emissions as a result of combined sewer overflow has in the past been tackled through an extensive programme implemented by Hamburg Wasser. This reduces the overall emissions into the surface waters; the current increase of intense rain however, is a factor that can in future result in again increased sewer overflows (Kopp, 2007).

Systems 5 BlaD and 6 CompU, which are based on the assumption that the current centralised wastewater system is replaced by decentralised systems, show greatly reduced nitrogen emissions to the environment. Sewer related emissions (infiltration from sewers and sewer overflow) and discharges from the WWTP do not exist in these two systems. Differences in rainwater emissions in 5 BlaD and 6 CompU are also due to other changes in rainwater management like the use of rainwater to replenish process water.

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86 The results of the model for discharge parameters N, P and CSB of the WWTP show a general correlation to real data (e.g. Hamburg Wasser, 2008a). Discrepancies observed are usually due to variations in assumptions, such as not taking industry into account, incorrect nitrogen discharge assumed (13 $mgN_{tot}\ l^{-1}$ instead of the actual 15.4 $mgN_{tot}\ l^{-1}$) or variations in rainwater volume in different years.

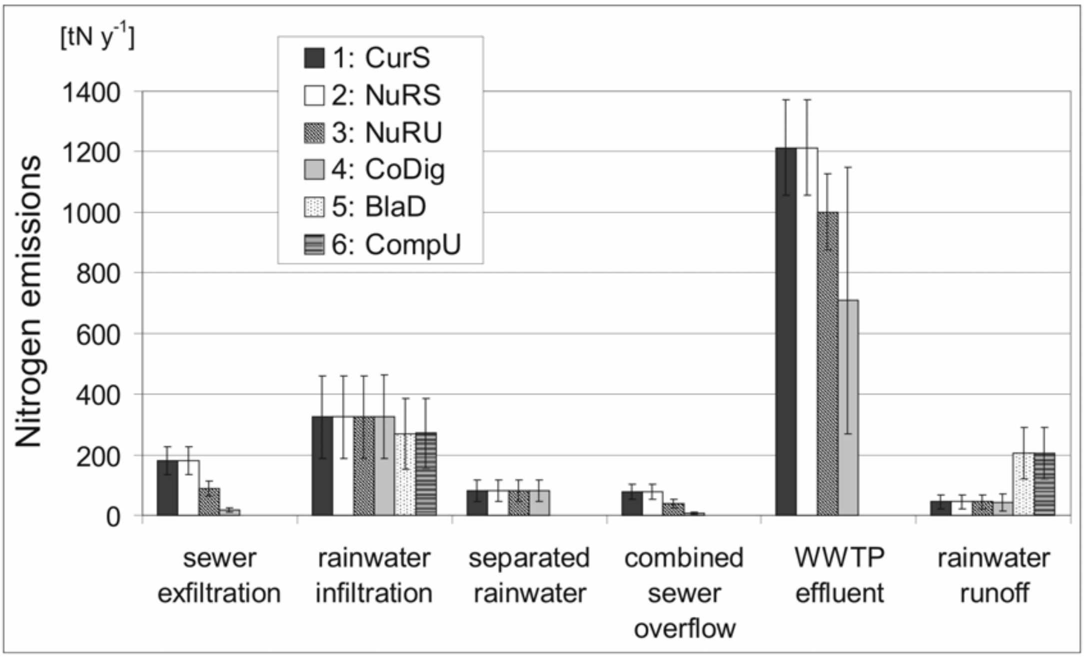

Figure 4.9: Total nitrogen emissions to soil/groundwater and surface water from water-system related sources [t_N y^{-1}]

Comparing the pollution from the urban wastewater system to that from agriculture, the large contribution of agricultural nitrogen discharges (e.g. from runoff or erosion) to the overall nitrogen emissions becomes apparent. According to the results of the ceMFA model, the nitrogen infiltration into soil from the total agricultural area amounts to 6200±2300 t_N y^{-1}, and the nitrogen discharge into surface waters from runoff and erosion of farmland is about 4300±1600 t_N y^{-1}. When considering only the administrative borders of Hamburg (i.e. without the hinterland area), agricultural emissions from infiltration and erosion/runoff contribute 235±87 t_N y^{-1} and 163±61 t_N y^{-1} respectively.

Table 4.5: Total nitrogen emissions to soil/groundwater and surface water from processes related to the urban water system [t_N y^{-1}]

	1: CurS	2: NuRS	3: NuRU	4: CoDig	5: BlaD	6: CompU
Emissions to soil/ groundwater	507 *±196*	507 *±196*	415 *±175*	344 *±157*	270 *±132*	271 *±132*
Emissions to surface water	1418 *±245*	1418 *±245*	1167 *±203*	841 *±514*	206 *±90*	206 *±90*

Nitrogen emissions to air can occur in processes such as composting, volatilisation from fertilisation or from wastewater flows, incineration of sewage sludge, and denitrification of nitrates in the wastewater treatment plant. The latter results in emissions of nitrogen gas (N_2), which enters again the nitrogen cycle and does not have any negative impact on the environment. Incineration and composting processes can however, result in the emission of nitrogen in the form of nitrogen oxides, which do have an environmental impact in terms of greenhouse gas emissions and acid rain. Nitrogen oxide emissions can be reduced by technical measures. These considerations however, go beyond the scope of this thesis, since no differentiation regarding the form of nitrogen is made, but total nitrogen is referred to.

The error margins of the results are relatively variable; this is particularly so for the nitrogen discharge from the wastewater treatment plant in System 4 CoDig, which seems rather variable (62% relative error). A sensitivity analysis reveals that a variety of parameters have an impact on this value. Among them are parameters such as nitrogen incorporation into biomass in the WWTP, the degree of nitrogen recovery in the stripping process of the WWTP (nutrient recovery process), the nitrogen content in greywater, and also the nitrogen content in organic waste. The latter is particular to this System 4, since organic waste is added to the digestion process, thereby increasing the nitrogen load in the activated sludge process because of recirculation of sludge liquor.

Phosphorus

The distribution of phosphorus emissions for the different systems (Figure 4.10) is similar to that of nitrogen emissions. As for nitrogen, discharges of phosphorus from the effluent of the wastewater treatment plant contribute the largest fraction of emissions for the Systems 1 to 4 (about 76-81% of the total discharges to surface waters). Source separation of urine or blackwater can reduce the WWTP discharges by about 15-17%. Discharges from combined sewer overflow can be reduced by about 32% (3 NuRU) or 75% (4 CoDig) by source separation. The total phosphorus emissions, which are shown in Table 4.6, are in the range of 33 to 70 $t_P\ y^{-1}$ and 25 to 122 $t_P\ y^{-1}$ for soil/groundwater and surface waters, respectively.

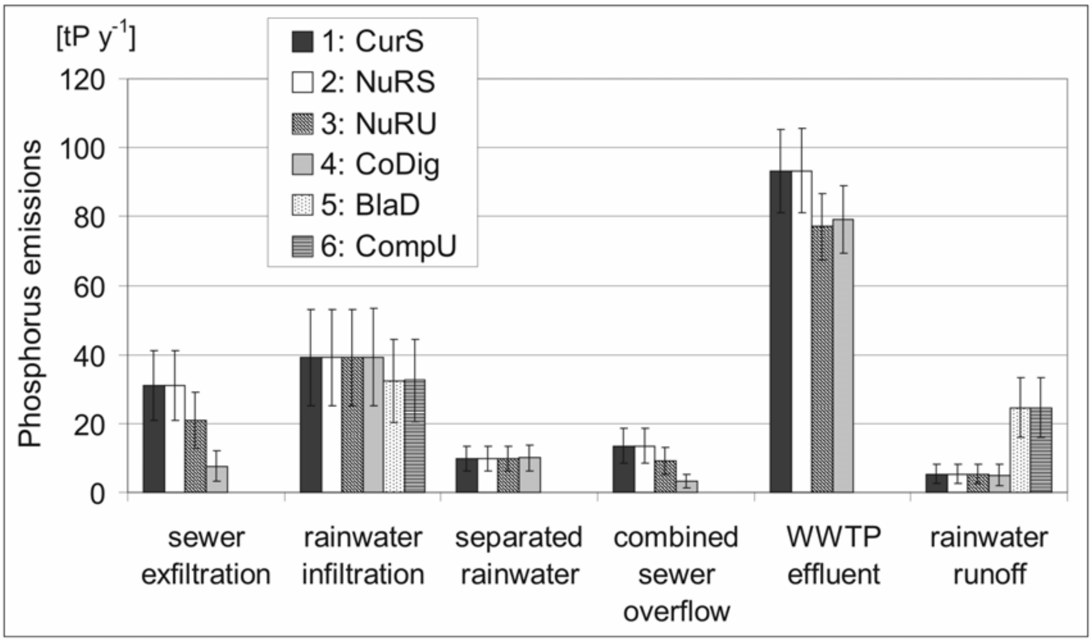

Figure 4.10: Phosphorus emissions to soil/groundwater and surface water from the different sources [$t_P\ y^{-1}$]

Erosion and runoff from farmland contributes about 68±25 t_P y^{-1} to diffuse phosphorus discharges to surface waters. Referring only to the agricultural area within Hamburg's city boundaries, this value is 3±1 t_P y^{-1}. Despite phosphorus being less mobile than nitrogen, there is some phosphorus infiltration, which amounts to 56±21 t_P y^{-1} for the total agricultural area and 2±1 t_P y^{-1} for the agricultural areas within Hamburg.

Table 4.6: Total phosphorus emissions to soil/groundwater and surface water from processes related to the urban water system [t_P y^{-1}]

	1: CurS	2: NuRS	3: NuRU	4: CoDig	5: BlaD	6: CompU
Emissions to soil/ groundwater	70 *±26*	70 *±26*	60 *±24*	47 *±20*	32 *±14*	33 *±14*
Emissions to surface water	122 *±24*	122 *±24*	101 *±20*	98 *±20*	25 *±9*	25 *±9*

4.2.2 Emissions of organic matter

The discharge of organic matter to surface waters can contribute to oxygen depletion and eutrophication of such water. The results of the modelling show the expected carbon discharges as total organic carbon (see Figure 4.11). In the systems 1 CurS and 3 NuRU the wastewater treatment plant discharges about 1750±419 t_{TOC} y^{-1}, which equals about 7000±1676 t_{CSB} y^{-1}. Source separation of urine in System 3 NuRU does not have any effect on the combined sewer overflows, since the depleted urine, which is still rich in organic carbon, is discharged into the sewer system. Discharge values of the WWTP can be reduced by source separation of urine or blackwater by about 15-17%. Emissions resulting from rainwater runoff contribute a large fraction of the total emissions (about 55-61% for Systems 1 to 4 and 100% for Systems 5 and 6). Yet, the high variability due to a high parameter uncertainty for TOC in rainwater runoff should be noted.

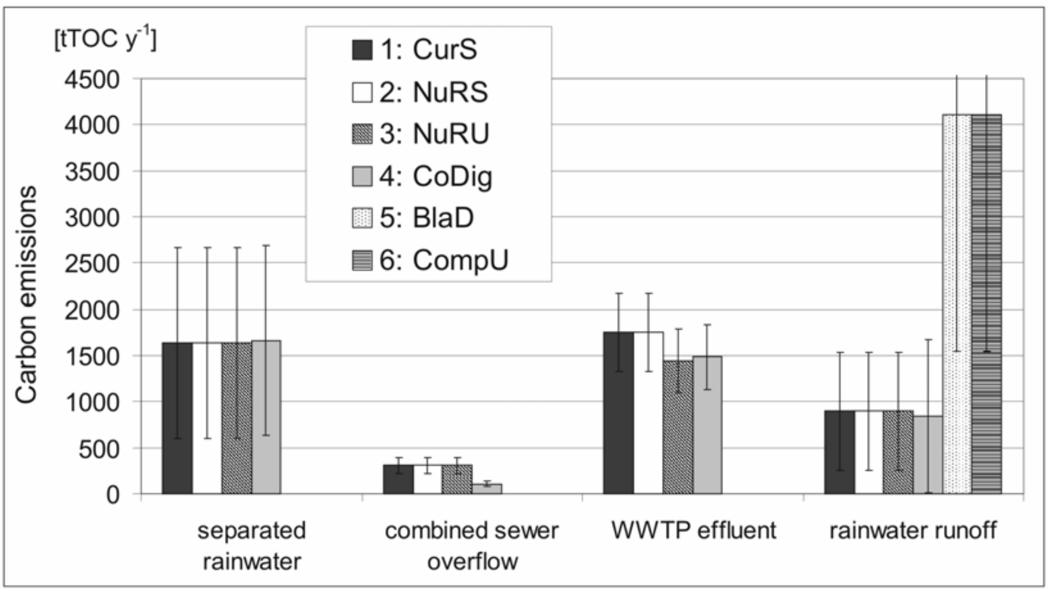

Figure 4.11: Carbon emissions to surface water from the different sources [t_{TOC} y^{-1}]

4.2.3 Nutrient recovery

One of the main aims of the five systems is the recovery of nutrients for fertilisation purposes. The results of the ceMFA modelling can be used to assess the potential of mineral fertiliser replacement. The total nutrient loads that can be recycled back to agriculture, taking into account the parameters from Section 4.1, are shown in Table 4.7.

Table 4.7: Specific recoverable nutrient loads [kg p^{-1} y^{-1}]

	1: CurS	2: NuRS	3: NuRU	4: CoDig	5: BlaD	6: CompU
Nitrogen	0.11±*0.08*	1.17±*0.37*	2.26±*0.74*	3.09±*0.98*	4.23±*1.08*	3.58±*1.28*
Phosphorus	0.03±*0.02*	0.46±*0.19*	0.20±*0.09*	0.44±*0.15*	0.61±*0.22*	0.55±*0.28*
Potassium	0.06±*0.04*	0.06±*0.04*	-	-	1.20±*0.28*	1.10±*.039*
Sulphur	0.01±*0.01*	(1.06±*0.43)*	*(2.61±.0.90)*	(3.71±*1.22)*	0.08±*0.06*	0.29±*0.12*

- *not applicable* *(….) from external sources*

Figure 4.12 illustrates the farming areas that could possibly be fertilised by products recycled from waste and wastewater flows. These products are on the one hand untreated flows such as urine or blackwater, and on the other hand products from treatment processes, such as MAP, compost, NH_4-solution, etc.

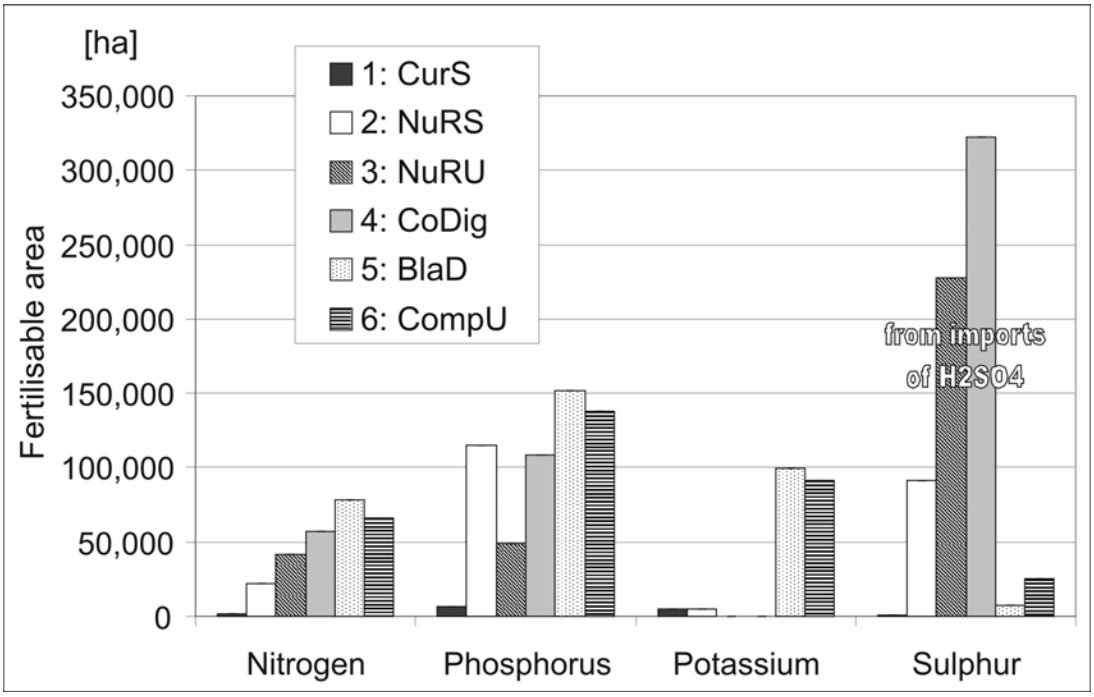

Figure 4.12: Nutrient recovery potential expressed as area that could be fertilised [ha]

Considering current nitrogen application rates, an area between 22,000 ha (2 NuRS) and 78,000 ha (5 BlaD) could be fertilised with recovered nitrogen. In the case of phosphorus, an area between 44,000 ha (3 NuRU) and135,000 ha (5 BlaD) could be fertilised with recovered phosphorus. System 3 NuRU is the only system that provides both nitrogen and phosphorus in a ratio similar to that of mineral fertiliser application rates, so that a total area of 42,000 ha to 44,000 ha could be exclusively fertilised with recovery products[87]. In all other system one nutrient is supplied in surplus and the area that could be fertilised with the recovery products differs for the particular nutrients. Considering the total agricultural area of about 500,000 ha (including the hinterland; see Section 4.1.1), the recycled flows cannot replace 100% of any of the studied nutrients. By contrast, if only the agricultural area within Hamburg's city boundaries is considered (19,200 ha), systems 2 to 6 provide all the required nutrients in excess amounts, so that export to the surrounding areas is possible.

The recovery potential depends on the nutrient under consideration and the kind of source separation and treatment processes that are applied in the wastewater system. The potential for mineral nitrogen fertiliser replacement can be up to 15% referring to the agricultural area in Hamburg and the hinterland. The highest recovery is achieved in System 5 BlaD, which includes the application of digested and sanitised blackwater on farmland. Also the other source-separating systems contribute to the recycling of nitrogen within a similar range. Regarding phosphorus, systems 5 BlaD and 6 CompU seem to be the most promising options with replacement rates of 29% and 26% respectively. Additionally, the recovery of MAP from sewage sludge (2 NuRS) or from sludge liquor (4 CoDig) can provide about 20% of the required phosphorus. Potassium can only be recovered in considerable amounts in Systems 5 BlaD and 6 CompU, which do not involve any treatment (in terms of recovery) of the source separated flows. The replacement rate in these two systems is about 15 to 17%. Sulphur replacement seems high in Systems 2 to 4, but this sulphur is actually imported into the systems as additives for the nutrient recovery processes. Therefore, it should not be considered when looking at recovery potentials. Thus, only systems 6 CompU and 5 BlaD provide a real sulphur yield with replacement rates of about 16% and 5% respectively. The application of compost from organic matter, as included in System 1 CurS, can only provide very low amounts of nutrients to agriculture (see also Table 4.7). Compost is a slow-release fertiliser providing plant nutrients over a longer time period. This aspect is however not included in the modelling; only yearly nutrient loads are looked at.

[87] No potassium and excess sulphur would however, be provided to that area.

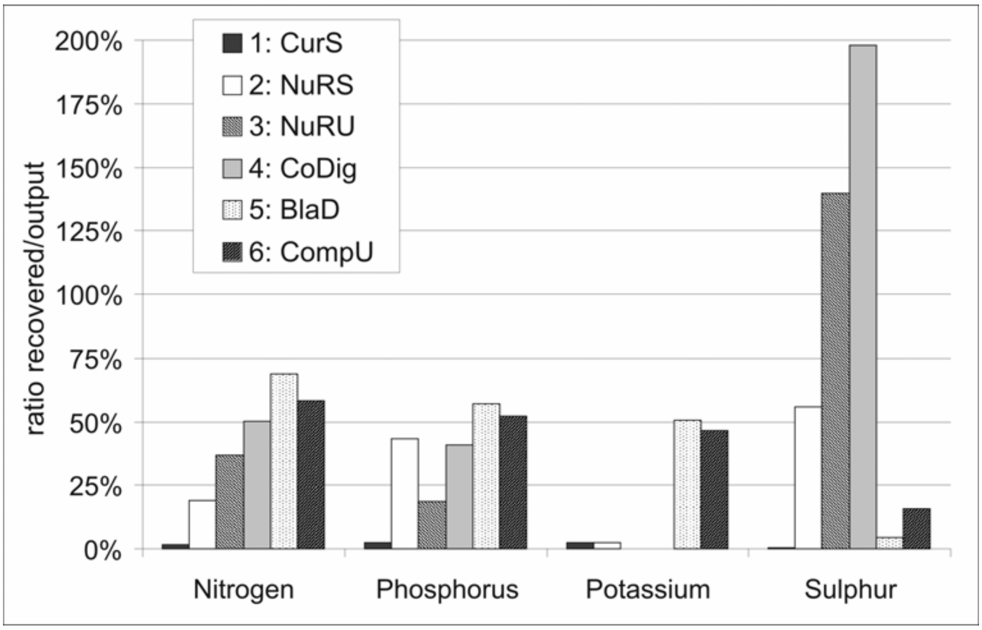

Figure 4.13: Ratio of recovered nutrient loads to nutrient outputs in urine, faeces, greywater and organic waste

In order to show the effectiveness of the different systems the ratio of recovered nutrients to the maximum available nutrients in the considered waste and wastewater flows, is expressed as a percentage. This is illustrated in Figure 4.13. Since the ratio of sulphur recovery is greater than 100% in Systems 3 and 4, it is apparent that sulphur recovery is mainly from other sulphur sources, and not from the domestic waste flows under consideration. Due to losses within the processes (i.e. ammonia volatilisation, collection rate, treatment efficiency, etc.) the recovery ratio for all nutrients in any of the considered systems is less than 100%. Particularly System 5 BlaD and System 6 CompU achieve an acceptable recovery of all nutrients, with recovery ratios for nitrogen, phosphorus and potassium, all in the range of 50% to 70%. The other systems have a recovery of nitrogen and/or phosphorus in the range of 37% to 50%.

4.2.4 Mass and water flows

The model results for mass flows can be used to study water flows such as the extraction of groundwater for drinking water provision. In addition, operational data such as the inflow of wastewater to the wastewater treatment can be illustrated. Figure 4.14 highlights these two water flows. It should be noted that the systems 5 BlaD and 6 CompU are based on the assumption that there is no centralised water supply. However, bottled drinking water is used for drinking water purposes in System 5; this flow, which is also shown in Figure 4.14, amounts to $1.3 \pm 0.2 * 10^6$ m^3 y^{-1}. It can be seen that groundwater extraction for domestic drinking water supply can be reduced by about 22-25 % through the introduction of source separation of urine or the use of vacuum toilets. Systems 5 and 6 do not require groundwater use. However, groundwater will be required for industrial purposes or for fire-fighting. It should be noted that reduced groundwater extraction has an impact not only on the natural water balance, but also on operational requirements of the water supply system. For example, low water flow and oversized pipes can require regular flushing of the pipes to prevent deposits and bacterial regrowth.

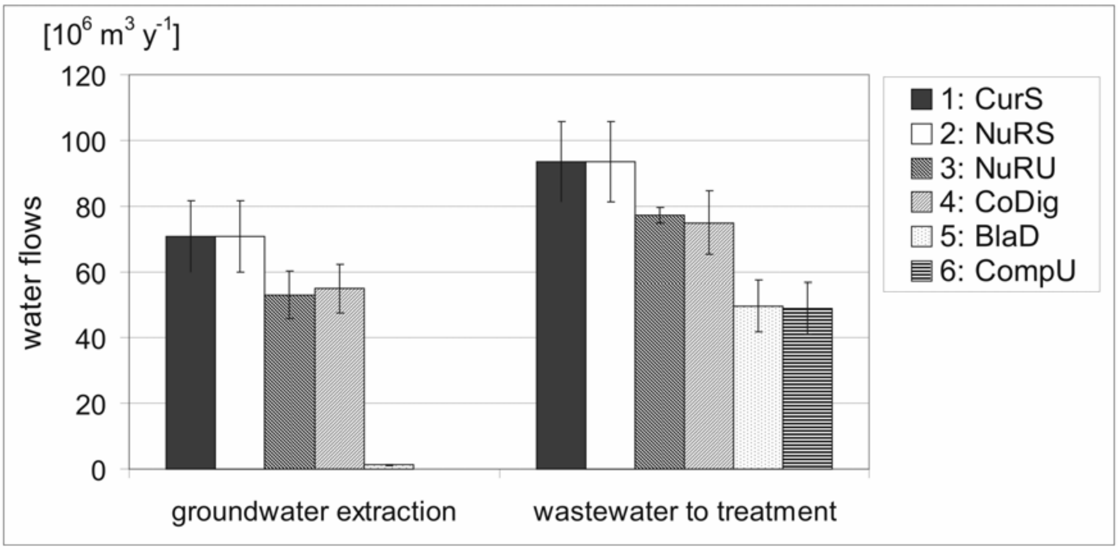

Figure 4.14: Groundwater extraction and inflow to wastewater treatment facilities [10^6 m^3 y^{-1}]

Another important result of the mass flow modelling is the calculation of lorry-based transport requirements as shown in Table 4.8. In systems 1 to 4 organic waste is collected from the households and transported to composting or anaerobic digestion facilities. Based on the assumptions shown in Section 4.1.2, this volume is annually about 38,000±18,000 tonnes. In these four systems the volumes of the products from human waste that are recycled back to agriculture[88], vary between 20,000±6,000 and 70,000±24,000 tonnes per year. The results of the modelling also show the volumes of source separated wastewater flows that need to be transported (e.g. by lorries) either to treatment facilities or directly to agriculture. Source separated urine (systems 3 NuRU and 6 CompU) amounts to 0.57±0.17 million tons per year. Blackwater (systems 4 CoDig and 5 BlaD) constitutes an even larger mass flow with 4.72±1.98 million tons per year. In comparison, the mass flow of mineral fertiliser is in the range of about 0.78±0.1 million tons per year. The calculated energy requirement based on mass flows and transport distances[89] is shown in Section 4.2.5.

[88] i.e. compost, MAP, phosphoric acid and ammonia sulphate

[89] For the calculation of transport distances see Section 4.1.7.

Table 4.8: Mass flows requiring lorry-based transport [1,000 t y^{-1}]

	1: CurS	2: NuRS	3: NuRU	4: CoDig	5: BlaD	6: CompU
organic waste to treatment	38 *±18*	38 *±18*	38 *±18*	38 *±18*	-	-
human waste to treatment	-	-	565 *±173*	4716 *±1975*	-	-
waste products to agriculture	28 *±13*	45 *±18*	20 *±6*	70 *±24*	4754 *±1976*	698 *±207*

\- *not applicable*

4.2.5 Energy analysis

Energy is required for a variety of purposes such as transport, water and wastewater treatment and for processes related to the recovery of nutrients. The total specific energy demand of every system (expressed as primary energy) and the contribution of different processes is shown in Figure 4.15 and Table 4.9. Compared to the current situation, only System 3 NuRU consumes considerably less energy overall (minus 12%). Considering all processes, System 3 requires 963±337 GWh y^{-1}, whereas Systems 1, 2, 4 and 6 all show similar energy demands in the range of 1080 and 1170 GWh y^{-1}. System 5 requires significantly more energy (plus 51%) with 1530±500 GWh y^{-1} in total. This is particularly due to the high energy demand for provision of bottled water. Also the processes wastewater treatment and nutrient recovery in this system (i.e. decentralised anaerobic digesters) show increased energy consumption.

The primary energy demand per capita varies between 553±128 kWh p^{-1} y^{-1} (System 3) and 876±147 kWh p^{-1} y^{-1} (System 5), including the production of mineral fertiliser. Referring only to the processes related to the water and wastewater system, the per capita primary energy demand ranges from 193 kWh p^{-1} y^{-1} to 547 kWh p^{-1} y^{-1}. This is less than 1.2% of the overall primary energy consumption per capita in Germany of 47,268 kWh p^{-1} y^{-1} (AG Energiebilanzen, 2009).

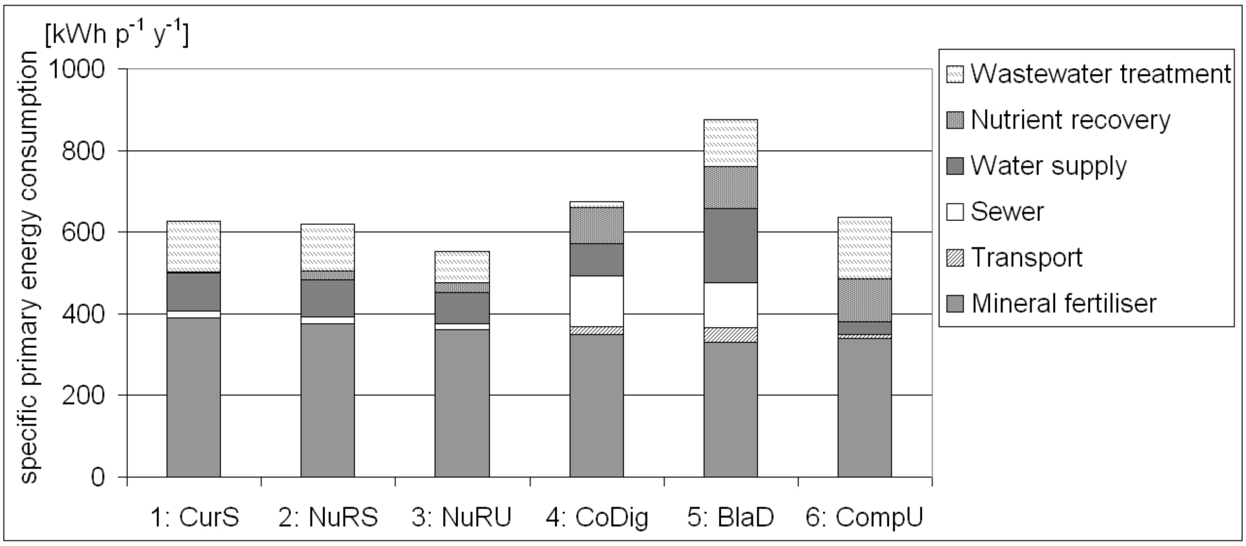

Figure 4.15: Specific primary energy consumption of the six systems [kWh p^{-1} y^{-1}]

It is apparent that the production of mineral fertiliser, which varies between 330 and 390 kWh p^{-1} y^{-1} depending on the replacement rate, constitutes the highest overall energy consuming process (38% to 65%). Although the process itself, i.e. production of mineral fertiliser, is not within the system boundaries, it is considered important to include the energy savings that can be achieved by the saving of mineral fertilisers. The second most important energy consuming process is the treatment of wastewater, either in a centralised plant (Systems 1 to 4), or the treatment of greywater in decentralised units (systems 5 and 6). Although the volume to be treated in the latter two systems is less than in the centralised systems, the energy demand is equal or higher in these two systems due to the higher treatment level for recycling purposes.

Systems 3 NuRU and 4 CoDig result in considerable energy savings in the wastewater treatment plant, compared to the current situation (1 CurS); this is due to the source separation of urine and blackwater[90]. Adding the separated blackwater to the anaerobic digesters in order to generate biogas instead of treating it aerobically (4 CoDig), can reduce the overall energy demand of the WWTP by 88% (see also Table 4.9). The nutrient recovery processes included in the systems, partly use up the energy savings. But even comparing the sum of energy demands for wastewater treatment and nutrient recovery of System 4 CoDig to the current situation, would still result in an energy saving of 18%. Energy consumption for nutrient recovery processes are particularly high where heating of blackwater and organic waste is included (5 BlaD), where large

[90] This is particularly due to the reduced need for nutrient elimination. Factors such as improved methane yield, resulting from urine separation as described by Wilsenach and van Loosdrecht (2003), are not included in the black-box model. Therefore, even higher energy benefits than described here, can be expected.

volumes of wastewater are subjected to recovery processes such as stripping (4 CoDig), or where decentralised ventilation is required (6 CompU).

Table 4.9: Primary energy demand per capita for the different processes [kWh p^{-1} y^{-1}]

	1: CurS	2: NuRS	3: NuRU	4: CoDig	5: BlaD	6: CompU
Transport						
Lorry	0.5 *±0.2*	0.5 *±0.3*	1.8 *±0.6*	20.6 *±5.7*	25.8 *±13.4*	9.1 *±4.6*
Conventional sewer	17.4 *±3.4*	17.4 *±3.4*	14.4 *±2.7*	13.9 *±2.7*	-	-
Vacuum sewer	-	-	-	109.7 *±42.0*	109.7 *±42.0*	-
Drinking water						
Centralised water supply	59.3 *±11.7*	59.3 *±11.7*	44.4 *±8.2*	46.0 *±8.4*	-	-
Bottled water	32.1 *±7.9*	32.1 *±7.9*	32.1 *±7.9*	32.1 *±7.9*	182.5 *±46.8*	32.1 *±7.9*
Wastewater/greywater treatment						
COD	28.4 *±10.8*	28.4 *±10.8*	29.0 *±10.7*	6.9 *±2.5*	-	-
N	44.1 *±16.2*	33.0 *±14.1*	19.1 *±8.6*	0 *±0*	-	-
P	2.3 *±1.5*	2.3 *±1.5*	1.2 *±1.1*	0.54 *±0.6*	-	-
Sludge treatment consumption	121.5 *±33.8*	109.4 *±30.2*	122.5 *±34.5*	162.5 *±95.6*	-	-
Sludge treatment production	129.5 *±37.0*	117.2 *±33.4*	143.4 *±40.7*	202.2 *±71.8*	-	-
Other (infrastructure)	58.3 *±23.8*	58.3 *±23.8*	48.2 *±19.6*	46.7 *±19.0*	-	-
Sum WWTP[91]	125.1 *±40.2*	114.1 *±30.4*	76.4 *±36.1*	14.4 *±62.6*	115.3 *±21.4*	151.8 *±28.2*
Nutrient recovery	-	21.1 *±6.5*	23.8 *±8.37*	88.6 *±30.2*	103.8 *±43.3*	105.0 *±12.6*
Mineral fertiliser	390.0 *±129.3*	374.5 *±128.4*	360.3 *±127.9*	348.5 *±127.6*	330.0 *±126.8*	339.0 *±127.2*
Total	626.9 *±129.1*	619.1 *±128.9*	553.2 *±127.9*	673.8 *±144.7*	876.7 *±146.8*	636.9 *±126.2*

[91] For systems 5 and 6 this value includes only the treatment of greywater in decentralised units. The treatment of blackwater or faeces/urine is included in the parameter “nutrient recovery”.

The introduction of vacuum sewers (as in systems 4 and 5) consumes considerable amounts of energy compared to conventional sewers. But on the other hand, energy requirements for water supply are reduced in systems that include low-flush or no-flush toilets. Lorry-based transport plays a role only in systems where blackwater is transported to an agricultural application or to treatment processes. In these cases, about 3% of the total energy consumption is due to fuel consumption of lorries[92]. By contrast, the transport of urine and compost seem rather insignificant. Comparing the energy required for transporting the untreated urine to an agricultural application, with the energy required for urine treatment (i.e. reducing the volume of the nutrients through MAP precipitation and stripping) reveals that the transport of untreated urine is more advantageous if the distance between the place of storage and the agricultural application is less than 110 km one way. However, these calculations do not include any considerations regarding improvements in energy efficiency of the urine treatment process itself. For example, improvements by introducing heat exchangers could be feasible. If the energy requirements for transport of untreated urine are compared with the energy savings for nitrogen removal on the WWTP, a maximum transport distance to an agricultural application of about 90 km, could be offset by the energy savings.

Drinking water consumption and the corresponding energy demand can be reduced considerably in a centralised system by the introduction of blackwater separation (4 CoDig). However, if water recycling is supposed to be complemented by provision of bottled drinking water (as in System 5 BlaD), the additional energy consumption of about 180 kWh $p^{-1} y^{-1}$ is rather high.

The specific energy demand for the decentralised treatment of greywater (5 BlaD and 6 CompU) is in the range of the current energy requirements for the centralised treatment of wastewater (115-152 kWh $p^{-1} y^{-1}$; compared to 125 kWh $p^{-1} y^{-1}$ in System 1). In addition, the energy demand for centralised drinking water supply of about 44 to 60 kWh $p^{-1} y^{-1}$ (depending on water consumption) is offset when looking at greywater recycling.

4.2.6 Economic evaluation

Although cost should not be the major criterion for efficiency assessments, it often plays a critical role in decision-making. The economic evaluation of the different systems is based on the process costs as listed in the process descriptions. The cost equations are

[92] Lorries for blackwater collection could be powered by refined biogas. This is already implemented in several case studies (Pölz and Salchenegger, 2005). With this measure increased greenhouse gas emissions and air pollutant emissions, due to the increased traffic, could be partly offset.

integrated into the material flow model and coupled to the relevant mass, nutrient or energy flows. Discounted investment costs as well as operation costs are included, added up and converted to annualised costs. Investment costs are discounted using an interest rate of 3% and the specific lifespans listed in the process descriptions. Operation costs include maintenance, servicing and repairs, which are partly calculated as percentages of investment costs and partly based on specific direct data. In addition, energy (i.e. electricity and heat) and material requirements are integrated into the cost calculations.

Costs of fertiliser are not included in the analysis. This is due to the fact, that benefits from selling recycled products, are later included in the evaluation.

The cost analysis is done for two different setups. Firstly, costs are calculated based on the existing infrastructure in Hamburg (Setup 1). In this case only re-investments and operation costs are taken into account; no investments for units such as the sewer system, the wastewater treatment plant and house installations are accounted for. Secondly, the cost calculations are carried out for a greenfield setup (Setup 2), where investments need to be done for all processes. The total costs for these two setups are illustrated in Figure 4.16. Costs are given as specific annualised costs, i.e. in euro per person per year. The cost calculation shows that the introduction of resource recovery in systems 2 to 6 generally increases the costs. Regarding Setup 1 (existing infrastructure), total costs vary between 162 € p^{-1} y^{-1} and 345 € p^{-1} y^{-1}. Particularly systems 4 CoDig and 5 BlaD show increased costs compared to the current situation (plus 60% and plus 114% respectively). Systems 3 NuRU and 6 CompU also show a cost increase, which is about 36%.

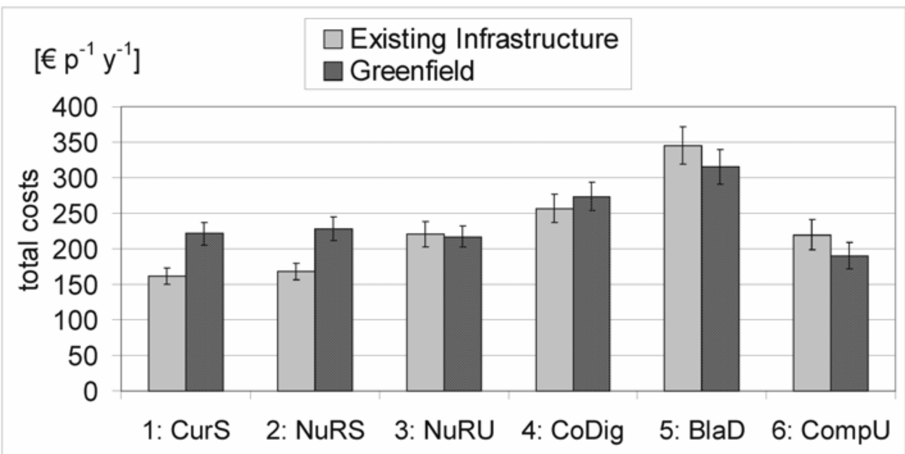

Figure 4.16: Total annualised cost per person including error margins [€ p^{-1} y^{-1}]

In the case of greenfield development, where all infrastructure needs to be built from scratch, costs for Systems 1 CurS and 2 NuRS increase to about 225 € p^{-1} y^{-1}. By contrast, specific costs for source-separating systems decrease in the greenfield case, since costs of retrofitting new toilets and piping are saved. In the greenfield setup, System 6

CompU is the most favourable cost-wise, with a specific cost of 190 € p^{-1} y^{-1}. Also the costs for System 3 NuRU are less than those for System 1 CuRS. The costs decrease for System 5 BlaD when compared to the case of existing infrastructure, since costs for retrofitting are saved. Despite this decrease in costs, it is still the most expensive system, at a cost of 315 € p^{-1} y^{-1}. In order to evaluate the differences between the systems, an overview of the overall cost breakdown is given in Figure 4.17 (existing infrastructure), Figure 4.18 (greenfield) and Table 4.10 (existing infrastructure).

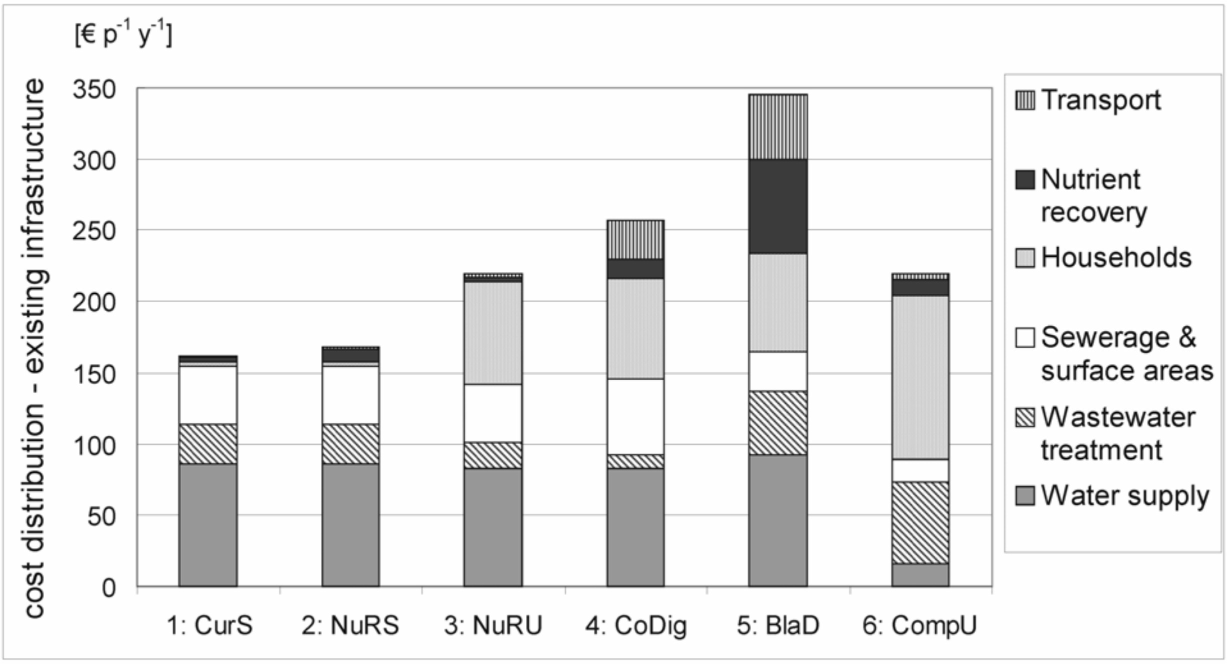

Figure 4.17: Breakdown of total annualised cost (existing infrastructure setup) [€ p^{-1} y^{-1}]

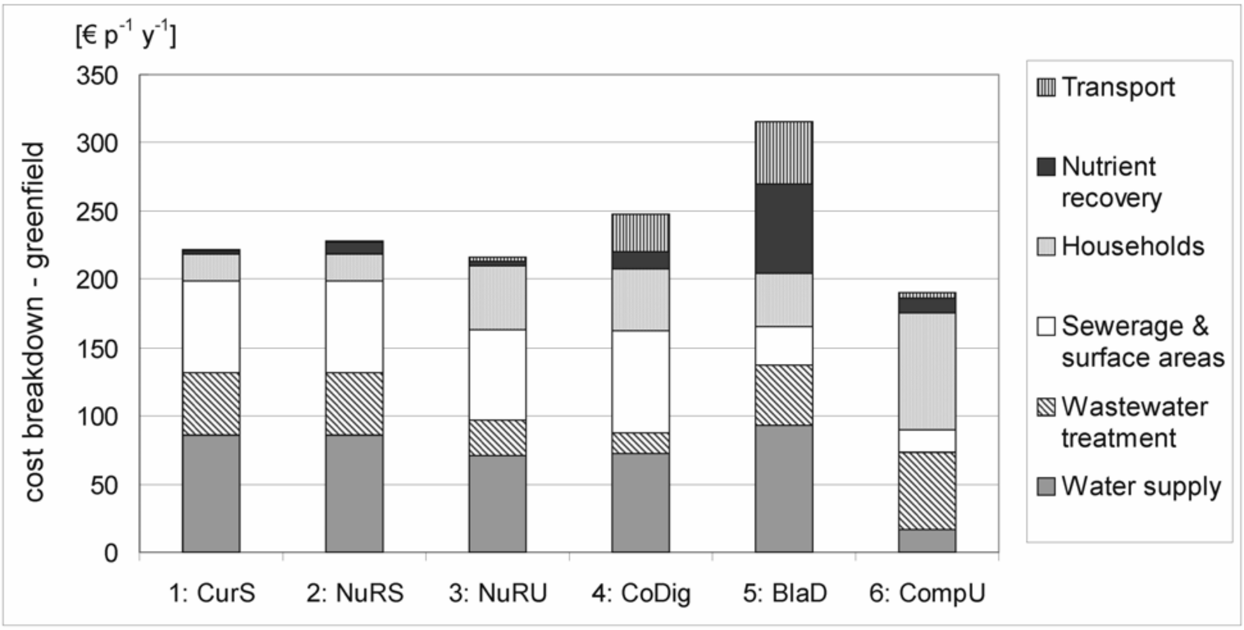

Figure 4.18: Breakdown of total annualised cost (greenfield setup) [€ p^{-1} y^{-1}]

In the centralised systems (systems 1 to 4), costs of the conventional sewer system make up about 16% to 20% of the total costs for Setup 1 (case of existing infrastructure). This value increases to about 30% to 37% for Setup 2 (greenfield case). Sewer costs are particularly high in System 4 CoDig, where additional vacuum sewers for blackwater collection need to be installed. Since the conventional sewer system is abandoned in Systems 5 and 6, costs for the Process Surface Areas and Sewerage decrease significantly; only costs for rainwater infiltration and for the vacuum sewerage in System 5 are included. In both systems (5 and 6) the centralised wastewater treatment plant is no longer budgeted for, but the cost for wastewater treatment does include the decentralised treatment of greywater in bio-membrane reactors, including installation costs. These costs are about 60% higher than the costs of wastewater treatment in the current situation. Comparing the decentralised treatment costs to the centralised treatment plant in the greenfield setup, shows that both cost factors are approximately the same if full investment costs are included (see Figure 4.18). Considering that in systems 5 and 6 the decentralised treatment also serves for water supply, highlights that decentralised greywater recycling can be competitive when compared to centralised systems particularly where there is no existing infrastructure. Costs of centralised wastewater treatment decrease noticeably if source separation is introduced (Systems 3 and 4). The reduced nutrient load results in cost savings for the activated sludge treatment processes. In addition, blackwater digestion as included in System 4 CoDig, can reduce energy requirements and contribute to cost savings.

The cost breakdown shows that transport contributes little to the overall costs. Only in Systems 4 and 5 (Setup 1), where untreated blackwater is transported, do transport costs make up more than 10% of the total costs. Nutrient recovery processes are also not one of the main cost fractions. In System 5 BlaD, where the anaerobic digestion process and the storage of slurry are counted as recovery processes, the costs contribute about 19% of the overall costs. Costs for households vary greatly across the different systems due to the replacement of the existing sanitary hardware and additional house connections. Annualised costs of source-separating toilets in Systems 3 to 6 contribute about 9% to 16% to the overall costs. Also operation and maintenance requirements add up to the households' costs. Households are by far the largest cost factor of System 6 CompU.

The breakdown of cost factors shows that with increasing decentralisation and source separation costs shift away from the centralised treatment processes towards the households. This emphasises the significant issue of cost allocation for project implementation, i.e. who is covering which costs. The difference between the two different investigated setups, i.e. existing infrastructure and greenfield, highlights the importance of properly defining the framework of the assessment. Existing infrastructure can hinder the introduction of new system components. Where the whole system is built from scratch, the total costs of the systems tend to be similar.

In order to evaluate the sensitivity of the results with respect to selected interest rates, a parameter variation is carried out. Figure 4.19 and Figure 4.20 illustrate the specific costs of the different systems at interest rates between 1% and 10%. For the case of existing infrastructure, Systems 1 and 2 do not show a significant dependency on the interest rate, since investments make up only a very small fraction of these systems. In general, higher interest rates lead to cost increases at comparable rates for Systems 3 to 5. Only costs for System 6 CompU, which requires larger investments in the case of existing infrastructure, increase to a greater extent than costs for the other systems.

Table 4.10: Costs of the different processes (case: existing infrastructure) [€ p^{-1} y^{-1}]

	1: CurS	2: NuRS	3: NuRU	4: CoDig	5: BlaD	6: CompU
water supply						
centralised	69.4	69.4	66.5	66.8	-	-
bottled water	16.3	16.3	16.3	16.3	92.7	16.3
transport						
organic waste	0.6	0.6	-	-	-	-
products (MAP, etc.)	0.3	0.5	1.1	1.5	-	2.4
urine	-	-	1.6	-	-	1.6
blackwater, slurry	-	-	-	26.0	45.8	-
wastewater treatment						
Q	4.8	4.8	4.0	3.9	-	-
COD	6.7	6.7	6.8	1.8	-	-
N	4.2	4.2	1.8	0	-	-
P	1.6	1.6	1.1	0.3	-	-
re-invest	10.9	10.9	4.5	3.3	-	-
MBR	-	-	-	-	44.6	57.3
organic waste treatment	2.7	2.7	1.4	1.4	-	-
households						
operation & maintenance	3.0	3.0	14.6	6.7	5.5	21.1
toilets	-	-	53.9	58.4	58.4	65.0
house connections, pipes	-	-	3.1	4.7	4.7	3.4
ventilation	-	-	-	-	-	25.2
storage (urine, blackwater)	-	-	0.2	0.8	8.0	0.9
sewerage						
operation & maintenance	41.0	41.0	40.8	42.1	2.0	-
vacuum sewerage incl energy	-	-	-	10.9	10.2	-
rainwater infiltration	-	-	-	-	15.9	15.9
nutrient recovery	-	6.0	2.9	12.1	57.6	10.4
total	161.6	167.8	220.7	257.0	345.3	219.5

\- *not applicable*

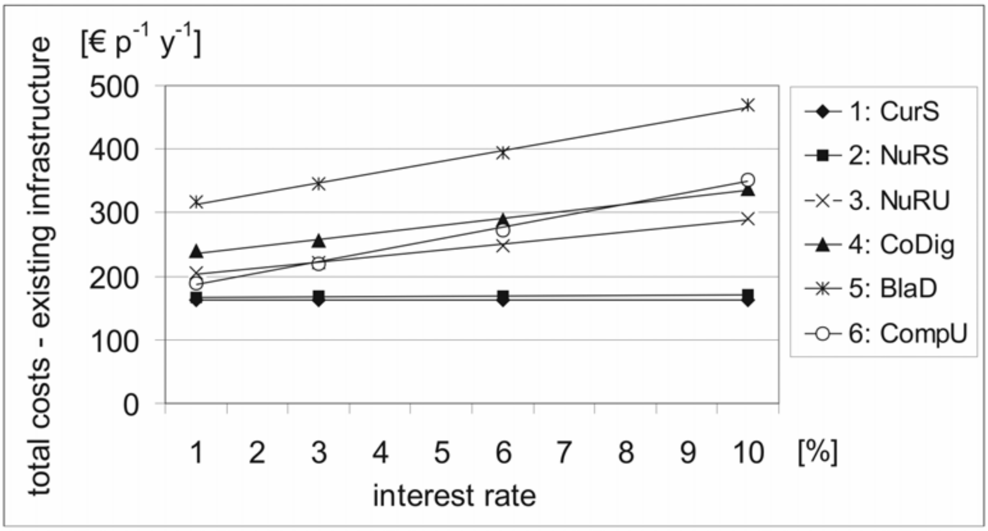

Figure 4.19: Specific costs at varying interest rates (existing infrastructure setup) (linearity assumed as approximation) [€ p^{-1} y^{-1}]

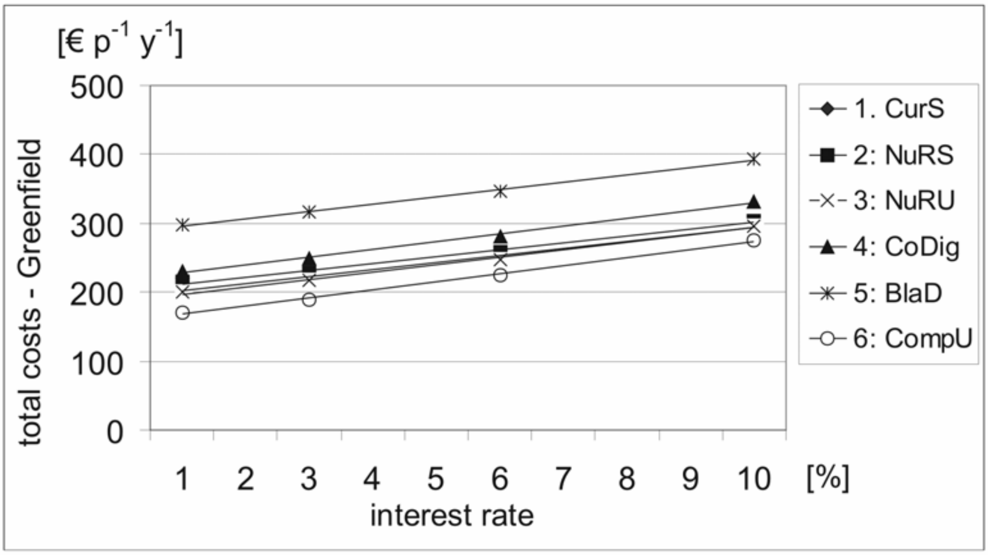

Figure 4.20: Specific costs at varying interest rates (greenfield setup) (linearity assumed as approximation) [€ p^{-1} y^{-1}]

Since one of the objectives of the different systems is the replacement of mineral fertiliser by recycled products, cost savings with regard to mineral fertiliser are expected. These benefits are accounted for in two different ways. Firstly, the calculation of potential benefits is carried out using only benefits that are currently achievable on the German market. This means, that products that are not yet available on the market, such as untreated urine, are not taken into account. On the other hand, any products such as compost, ammonia solution or MAP, which are already common merchandise, are considered using their commercial prices as indicated in the respective sections of Chapter 4.1. The second approach is a more optimistic calculation assuming that in future human excreta products will be valued according to their nutrient content and the corresponding monetary value of the nutrients. This calculation is in line with the work of other authors such as Dockhorn (2007). Using current fertiliser prices as a basis, the cost per tonne of fertiliser is converted into specific costs per nutrient element. This approach results in specific prices as listed in Table 4.11.

Table 4.11: Specific prices per nutrient element (based on Esemen and Dockhorn (2009), 2008 prices)

	N	P	K	S
Price per element [€ kg^{-1}]	1.03±0.10	3.13±0.30	0.60±0.06	0.36±0.04

The overall benefits per person per year calculated according to the two above mentioned approaches are illustrated in Figure 4.21. It is apparent that the approach of using nutrient specific prices, results in higher benefits than the calculation with current market prices for recycling products. Only in System 1 CurS, where just compost is sold, is the current benefit slightly higher than from the calculation with nutrient specific prices, because the latter approach neglects the value of organic matter. Considering System 5, where slurry is used in agriculture, current market conditions and prices paid for recycled products do not result in any benefits. However, if the equivalent value of the nutrients is taken into consideration, this system yields the highest benefits; about 7.0±1.6 € p^{-1} y^{-1}. This highlights the importance of getting recycled products from human waste and excreta, such as stored urine or digested slurry, established as accredited fertilisers; recognition of the fertilising value and the accompanying benefits will ensure that higher prices are achieved.

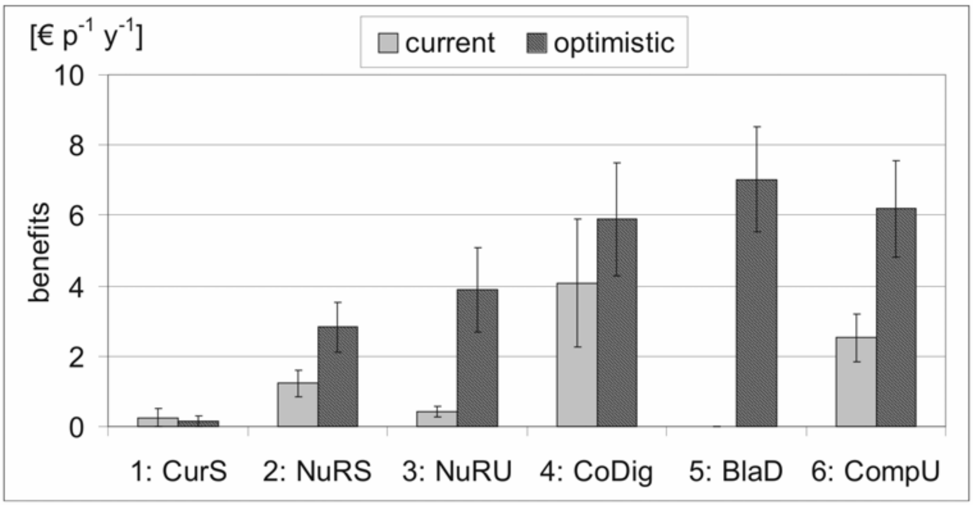

Figure 4.21: Specific benefits from nutrient recycling products including error margins [€ p^{-1} y^{-1}]

Comparing benefits and costs for nutrient recovery, only System 3 NuRU achieves benefits greater than the costs spent. If the nutrient specific values listed in Table 4.11 are taken into account, the overall benefits for System 3 amount to 6.8±2.1*10^{6} € y^{-1} compared to costs for urine treatment (MAP precipitation and steam stripping) of about 5.1±1.5*10^{6} € y^{-1}. However, if transport and storage costs are included, then the costs outweigh the benefits also in System 3. Therefore, based on the cost functions and current unit costs included in the model no overall financial benefits of nutrient recycling can be observed.

In economic analyses the parameter *benefit-cost ratio* is very often used for decision making. This parameter looks at the ratio of benefits to costs for a specific activity or a project, to assess whether the implementation of the project is worthwhile. Usually, only the costs directly related to the new implementation, i.e. the specific process, are considered. In this study, however, it is argued that the costs related to the whole system need to be included, since additional cost and benefits can occur within the whole system. Figure 4.22 illustrates the ratios of benefits from nutrient recycling calculated based on fertiliser equivalents (as shown in Table 4.11), compared to the annualised total system costs for the case of existing infrastructure. The values represent relative values, since costs occurring in all systems are not included. The higher the value of the ratio, the more beneficial a system is in terms of costs and benefits. System 6 CompU is the most advantageous one in terms of benefit-cost ratio, followed by System 4 CoDig. Although System 5 BlaD achieves the highest recovery rate of all systems, its high costs outweigh the benefits and result in a rather unfavourable benefit-cost ratio. All systems show significantly higher costs than benefits and have therefore very low benefit-cost ratios. This is due to the fact that the most important benefits of the systems, such as management of wastewater to prevent diseases and protect the environment, are not valued monetarily in this study as they represent intangible benefits.

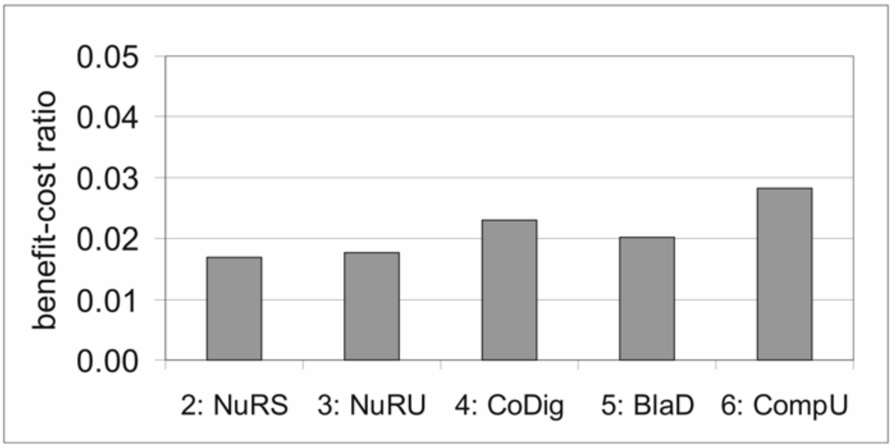

Figure 4.22: Benefit-cost ratios of Systems 2 to 6

If only additional costs of the systems compared to the current situation are looked at, the ratio of benefits to additional costs (i.e. benefit-Δcost ratio) of systems 3 to 6 varies between 0.02 and 0.06. Only System 2 NuRS shows a significantly increased benefit-cost ratio of 0.26 due to the comparatively small change in overall system costs. However, because this ratio is less than one, it also means that at current nutrient market prices and at the assumed costs for implementation of nutrient recovery, the system change in case of existing infrastructure is currently not beneficial from a monetary point of view. As discussed above, the situation is different for the case of new developments, i.e. greenfield developments, where systems 3 NuRU and 6 CompU are less costly than System 1 CurS and the benefit-Δcost ratios of the other systems are also more advantageous than when existing infrastructure is considered.

4.2.7 Discussion of the results

In this section a summary of the criteria nutrient recovery, water consumption, energy use and costs of the different systems is given (see also Figure 4.23). The main objective of the development of the five alternative systems (systems 2 to 6) is the recovery of nutrients. This is reflected by replacement rates[93] for mineral nitrogen or phosphorus fertiliser between 4% and 29%. In general, the replacement of phosphorus fertiliser is comparatively higher than the replacement of nitrogen fertiliser. Particularly System 2 NurS recovers a good deal more phosphorus than nitrogen. The recovery of one nutrient in surplus means that the other nutrients still need to be added in the form of balanced mineral fertiliser.

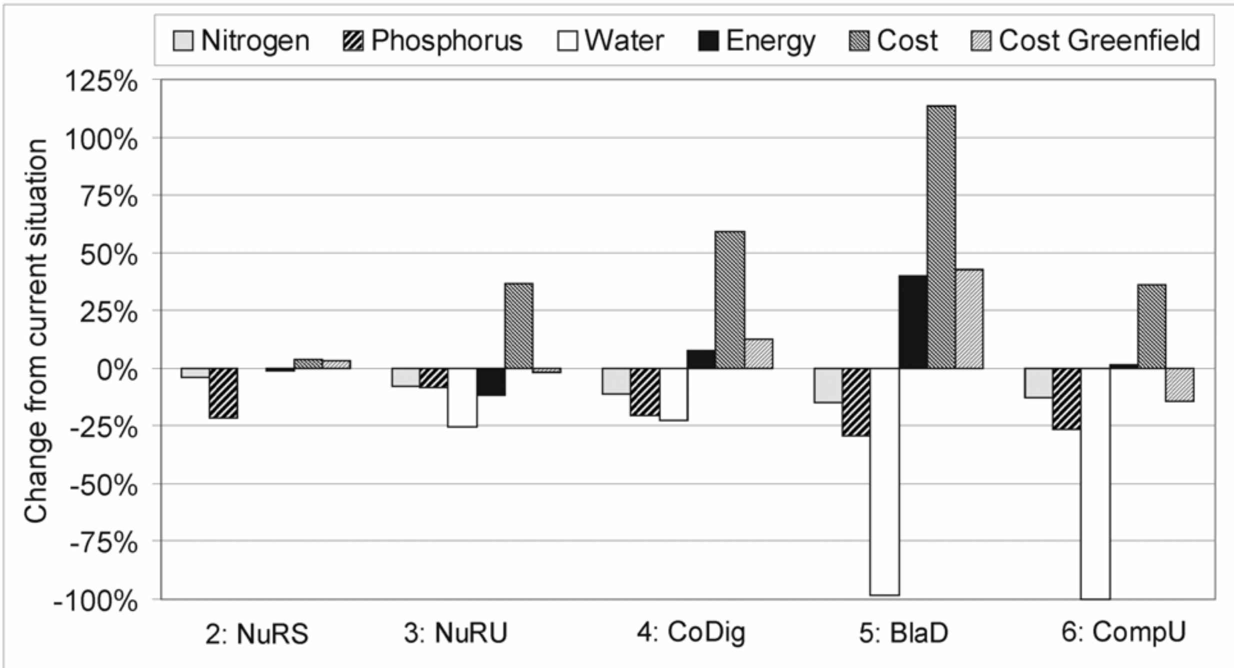

Figure 4.23: Potential change of nitrogen and phosphorus fertiliser use, groundwater extraction, energy demand and annualised costs compared to the current situation [%]

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[93] In this context it needs to be noted that the model uses average fertiliser application rates. In farming practice, however, nutrient application depends on specific crop and soil requirements.

The highest replacement of mineral fertiliser is achieved by System 5 BlaD with replacement rates of nitrogen and phosphorus fertiliser of about 15% and 29% respectively. However, the transport of digested slurry and particularly the use of bottled water, counteract these benefits and make this system the most unfavourable in terms of energy use (40% higher than currently) and costs (114% higher). In addition, biogas production from the current system setup seems to be too low to make up for increased energy demand for digestion and vacuum sewers. In this regard please refer to Section 4.3.3 for a discussion of significant parameters, which can have a positive effect on the efficiency of the system.

Water consumption can be reduced by source separation of urine or blackwater. System 3 NuRU and 4 CoDig reduce water consumption by about 20 to 25%, whereas the decentralised systems (5 BlaD and 6 CompU) require almost no external water sources due to the recycling of process water and the use of rainwater.

Overall energy demand is reduced by the introduction of source separation of urine (3 NuRU) or by nutrient recovery from sludge (2 NuRS). All other systems show higher energy needs. The relatively poor performance of systems 4 and 5 in particular is rather unexpected; these systems are characterised by the anaerobic digestion of waste flows and the related energy generation from biogas. The reason for this is the current system setups, including for example, the relatively high flush volumes of vacuum toilets and the relatively low amount of organic waste added to the digestion process. A parameter variation as carried out in Section 4.3.2 is therefore essential.

Costs for Systems 3 to 6 are significantly higher than costs for the current situation (see also Section 4.2.6). This picture changes in the case of greenfield development, i.e. if the use of already existing infrastructure is neglected. Then, overall costs for Systems 3 NuRU and 6 CompU are even less than for System 1 CurS, and also the relative costs for Systems 4 CoDig and 5 BlaD are greatly reduced. Considering the detailed cost breakdown, a reallocation of costs to the source (i.e. the households) can be observed by the introduction of source separation. This does not mean though, that the costs need to be actually borne by the households themselves. Cross-subsidies could for example be introduced.

The substitution of water from decentralised water supply systems by bottled water (as modelled in System 5 BlaD) is considered to be a very negative factor with regard to energy and cost balances[94]. If social acceptance allows, treatment levels for recycled

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[94] Bundanoon in Australia was the first community world-wide to ban bottled water from its shops to protest against the use of resources related to bottled water. On request, re-usable bottles are filled with tap water by local businesses (Wälterlin, 2009).

water should be high enough to ensure the provision of safe drinking water (as included in System 6 CompU). If the consumption of recycled water is not socially acceptable, the provision of drinking water can also be ensured by a centralised system. However, a dual system (centralised provision of drinking water and decentralised facilities for process water) seems to be economically inferior compared to one single system.

Although System 3 NuRU has one of the lowest overall fertiliser replacement rates (about 8% for both nitrogen and phosphorus), it shows the lowest overall energy demand; current energy use is reduced by 12%. Also, water extraction is reduced by more than 25%. It is therefore one of the more promising systems, particularly in the case of new developments (i.e. greenfield) where total costs are even less than those of System 1 CurS.

All in all, the analysis shows that none of the systems is superior in all criteria. Therefore, an aggregation of the criteria would be needed to come to a general ranking of the systems. However, such an aggregation requires the weighting of the criteria, which is beyond the scope of this study because stakeholders need to be involved in the weighting process. The depiction of the discrete results furthermore keeps the analysis transparent. Using multi-criteria decision support, such as a decision-matrix where the performance regarding a range of different criteria is assessed, would furthermore require that the criteria be independent[95].

Therefore, it is up to the planner or decision-maker to evaluate the results according to the specific needs. Trade-offs need to be taken into account as benefits related to some criteria are connected with drawbacks in other criteria. For decision making a transparent and well-coordinated process would be required.

It is important to note that modelling results always depend on the assumptions used for the model approach. Therefore, the results shown here should always be considered in the context of the system setup and the selected parameters. The next section (Section 4.3) details the analysis by varying system parameters and system setups, in order to assess the sensitivity and variability of the modelling results and to look for measures to improve the resource efficiency of the systems.

[95] This means that, for example, the replacement of nitrogen fertiliser, which is energy consuming process, as a target, must be separated from the energy balance to avoid any double counting.

4.3 Sensitivities, parameter variations and system modifications

The assessment in the previous sections is dependent on the specific selection of particular parameters used to define the system characteristics. Most of these parameters are intrinsic to specific processes or flows and well determinable, whereas some are selected by approximation. It is therefore interesting to check the dependency of the systems on the parameters and to identify key parameters, i.e. sensitive parameters (Section 4.3.1). Furthermore, the modelling setup allows the assessment of the impact of parameter variations on the results; this is discussed in Section 4.3.2. This gives insight into how the results are affected by changing parameters when assumptions are varied, and which system improvements can be gained by modifiable parameters[96]. In Section 4.3.3, some recommendations referring to the general set-up of the systems are given and possibilities for improvements are indicated.

4.3.1 Key parameters

A sensitivity analysis is used to identify key parameters (i.e. sensitive parameters) for specific criteria. Annex E highlights the ten key parameters for every system for each of the criteria: nutrient recovery, energy demand per person, and costs per person. Since the identified key parameters are particularly important for the results, their value should be selected with great care in order to get reliable results. In the following paragraphs, the key parameters are briefly discussed. More detailed results can be found in Annex E.

Regarding the amount of recycled nutrients to agriculture, the number of inhabitants plays a major role. This can also be transferred to the number of connected inhabitants in the case of a stepwise implementation. Furthermore, for the case of urine-separation toilets, the amount of collected urine, i.e. the separation effectiveness of the toilet, is very important. If the separation mechanism of the toilet does not allow a satisfactory collection of urine, this directly threatens the overall efficiency of the system. In other words, if only 35% of the urine is collected instead of the assumed 70%, the recovered nutrient loads will decrease by 50%. Other parameters with a relatively high sensitivity are the nutrient contents of the wastewater fractions. Considering the relatively high variability of these values, this emphasises the need for improved knowledge of the characteristics of source separated wastewater flows.

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[96] In this thesis, modifiable parameters in contrast to changing parameters are defined as those parameters that can be varied purposefully by respective system adaptations.

Since the total energy demand per person is a sum criterion, there is not one single parameter that shows a great sensitivity because several parameters impact on it. As energy demand for mineral fertiliser represents a large fraction of the overall energy demand, parameters that impact on the fertiliser-related energy demand are important; parameter such as food consumption per person, fertiliser application rates and energy consumption of fertiliser production. Also, the conversion factor of electricity to primary energy is relatively important. Thus more efficient electricity production could decrease the energy consumption in all systems. Another parameter showing up in the sensitivity list of all systems is the amount of greywater. Reducing the amount of greywater by 10%, would decrease the overall energy consumption by about 1.1% to 2.2%. Of relative importance for the centralised systems (1 to 4) is the energy demand for sludge drying, which is a high-energy consuming process. For Systems 4 CoDig and 5 BlaD the amount of flush water in the vacuum toilets and the electricity demand for vacuum sewerage are sensitive parameters. Therefore, technical improvements regarding the vacuum collection of blackwater, directly impact on the overall energy efficiency of the respective systems.

Regarding specific costs the cost parameters for water supply (for both centralised and bottled water supply) have a relatively high impact on the overall result. This includes costs per cubic metre, fixed costs and water consumption. Also sewer operation and maintenance is a sensitive parameter for the centralised systems (1 to 4). Since toilets need to be replaced in source separating systems (3 to 6), the average number of toilets per household, as well as the toilet costs and costs of retrofitting, are another group of parameters that have a relatively high importance for overall costs. Economies of scale can be observed by the impact of the parameter "number of inhabitants". A 10%-increase of the number of inhabitants would result in the specific costs to decrease by about 1.6% (5 BlaD) up to 4.8% (4 CoDig).

4.3.2 Variation of selected parameters

Since the equation system, and therefore the modelling results, depend on the selected parameters and the data quality, it is important to check the impact of varying parameters on the variables. One of the highly variable parameters is the nitrogen content in wastewater flows, particularly in urine (see also Section 2.4.2). Certainly, there is a direct dependency of the recycled nitrogen on the nitrogen content in urine[97] (see also above and Annex E). However, there is also a connection between the overall energy consumption and the parameter "nitrogen in urine". A parameter variation of

[97] For example, a 10% increase of the nitrogen load in urine increases the recovered nitrogen in systems 3 and 6 by 10%.

the specific nitrogen load excreted in urine (see Figure 4.24), shows that for Systems 2 NuRS and 4 CoDig the energy consumption increases with increasing nitrogen loads. This is particularly due to higher requirements for wastewater treatment and nitrogen recovery. On the contrary, for Systems 3 NuRU, 5 BlaD and 6 CompU, the overall energy demand decreases with increasing nitrogen loads. The impact of varying phosphorus loads in urine, however, is negligible.

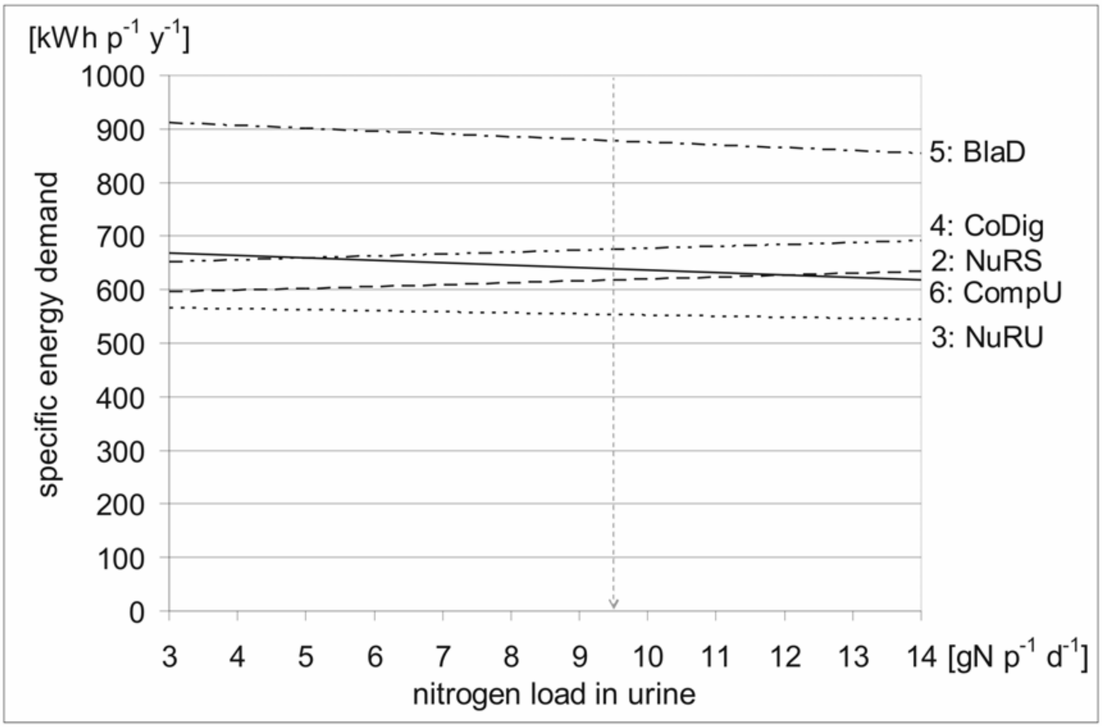

Figure 4.24: Impact of varying nitrogen loads in urine on specific energy demand (Systems 2 - 6)
(initial parameter marked in the figure)

To show how technical improvements could influence the efficiency of source separating systems, a parameter variation on the collection ratio of urine is carried out. This parameter is set to be at a relatively low 0.7 in the initial modelling. However, if the urine collection mechanism of toilets is not working properly and if people are not motivated, this value could even be lower. As mentioned above, there is a direct relationship between the nutrient recovery potential and the ratio of urine collection. However, the energy demand is also connected to the ratio of urine collection. As Figure 4.25 illustrates, System 3 NuRU shows a greater dependency on the collection ratio than System 6 CompU, where the volume of the urine is not reduced, and thus energy savings related to reduced mineral fertiliser consumption, are partly offset by increased transport requirements. For System 3 NuRU overall energy savings at a 100% urine collection would only amount to about 3%.

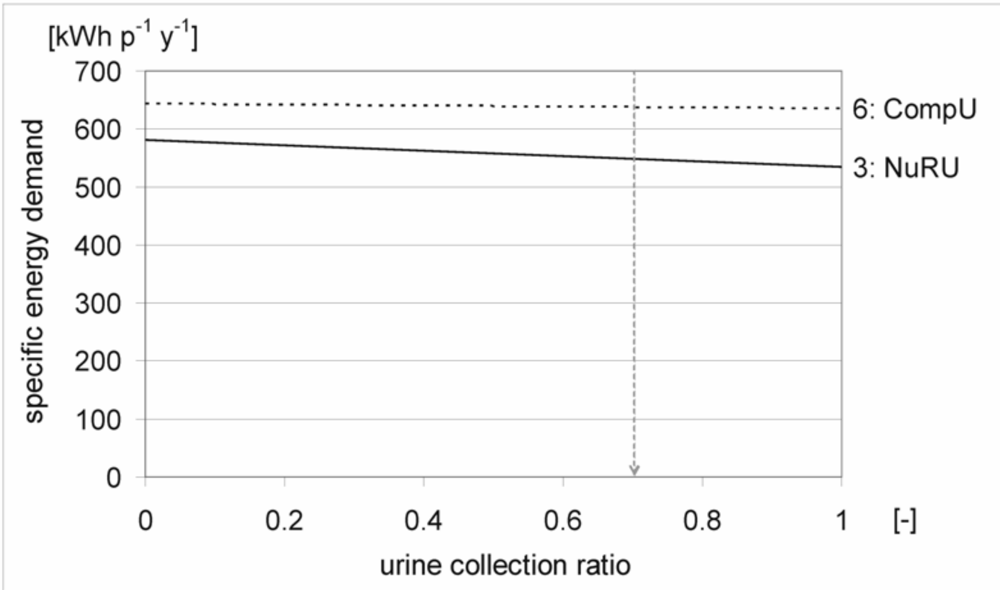

Figure 4.25: Impact of varying urine collection ratios on specific energy demand (Systems 3 NuRU and 6 CompU) (initial parameter marked in the figure)

For the modelling of the two systems (4 CoDig and 5 BlaD) based on vacuum toilets an average flush water consumption of 6 l p^{-1} d^{-1} is assumed. In order to see how this assumption affects the results, a parameter variation is carried out. According to Wendland (2008) vacuum toilets with a flush water volume of 0.25 l $flush^{-1}$ have been successfully implemented in airplanes. Therefore, an average daily flush water volume as low as 1.5 l p^{-1} d^{-1} could be realistic from a technical point of view. On the other hand, if the flushing procedure is not satisfactory, people will tend to flush several times, thereby increasing the amount of blackwater that needs to be treated and transported. Another important parameter that impacts on the result of the Systems 4 and 5 is the ratio of collected organic waste. The initial modelling assumes the same collection ratio as today, which is about 22%. It can however be assumed that the collection ratio will increase particularly due to expected energetic benefits resulting in a higher incentive for operators of the system.

Figure 4.26 to Figure 4.29 illustrate the resulting energy demand and costs of this parameter variation for systems 4 CoDig and 5 BlaD. It is apparent that for both systems an increased addition of organic waste increases the energy gains at slightly increasing costs. Energy consumption and costs are also directly dependent on the amount of flushwater. Particularly in System 5 BlaD reduced flush water considerably decreases the specific energy demand as well as costs; in this system blackwater is heated and digested in decentralised units and then transported to agriculture. Combining an increased organic waste collection and lower flush volumes can result in improved energy gains. For example, for System 4 CoDig a collection ratio of 0.8 and a daily flush water volume of 2 l p^{-1} d^{-1} brings down the specific energy demand to about

545 kWh p^{-1} y^{-1}. Thus, the energy consumption is comparable to System 3 NuRU and even less than the current energy demand in System 1 CurS. Costs amount to 256 € p^{-1} y^{-1}. For System 5 BlaD the same parameter setting results in a specific energy demand of 680 kWh p^{-1} y^{-1} and specific costs of 315 € p^{-1} y^{-1}. This emphasises the great dependency of the resource efficiency on certain parameters. Therefore, proper use and operation is crucial when it comes to implementation.

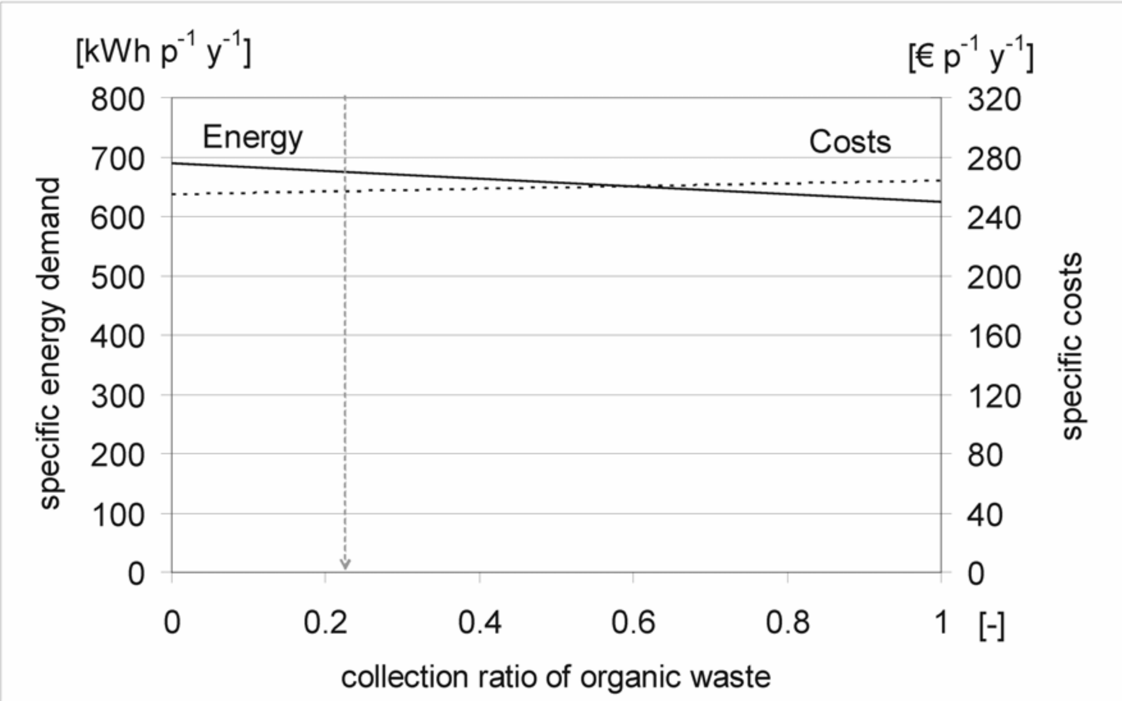

Figure 4.26: Impact of varying ratio of organic waste collection on specific energy demand and costs (System 4 CoDig)
(initial parameter marked in the figure)

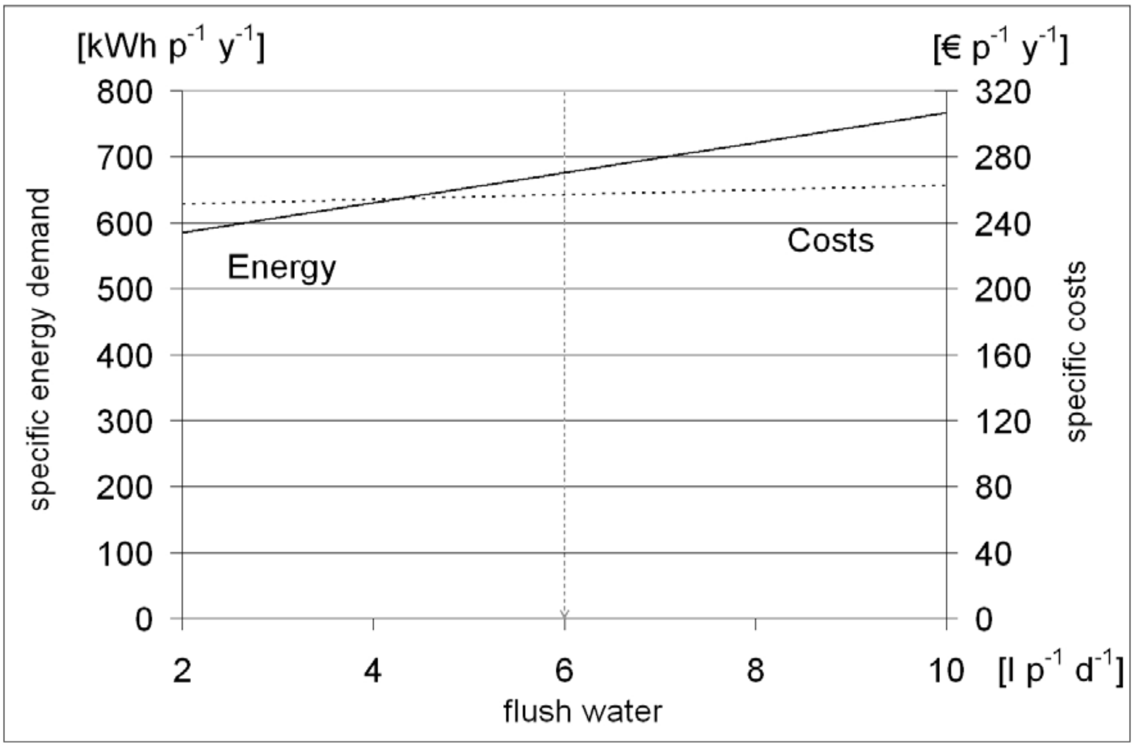

Figure 4.27: Impact of varying flush water volume on specific energy demand and costs (System 4 CoDig)
(initial parameter marked in the figure)

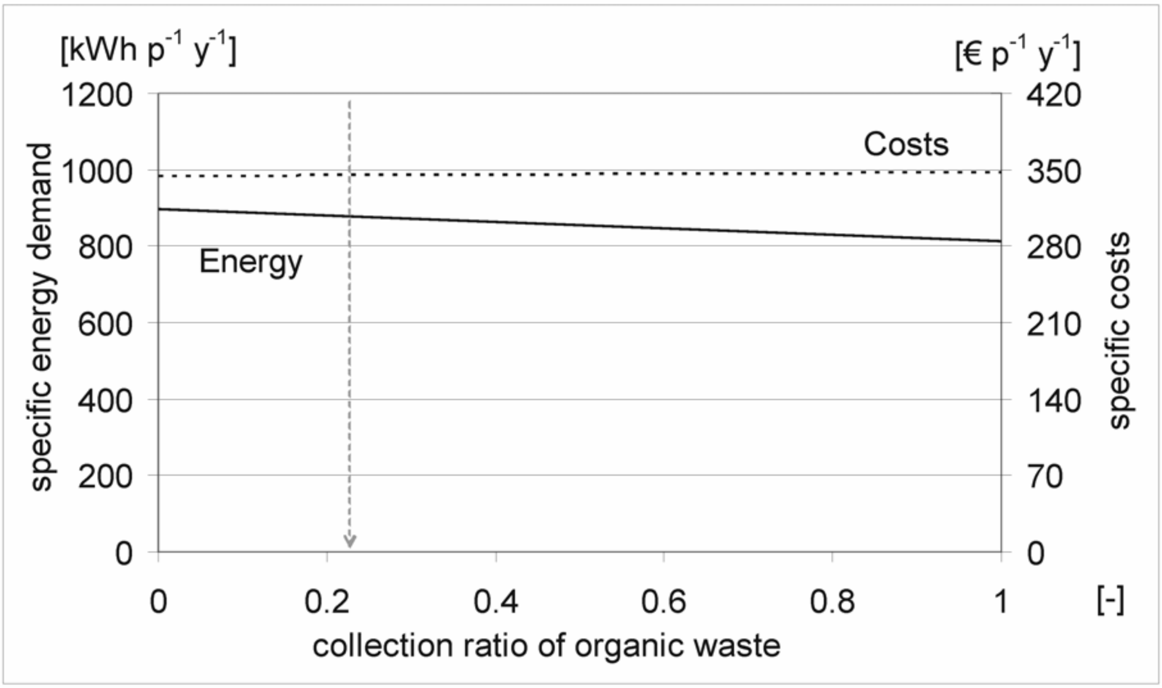

Figure 4.28: Impact of varying ratio of organic waste collection on specific energy demand and costs (System 5 BlaD)
(initial parameter marked in the figure)

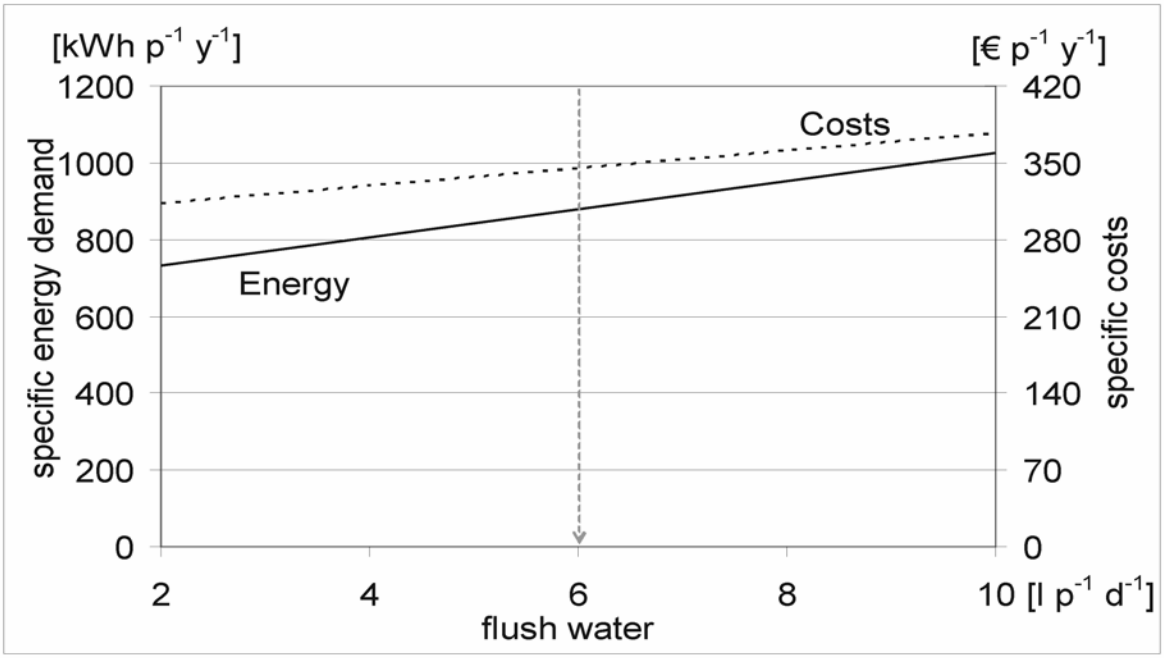

Figure 4.29: Impact of varying flush water volume on specific energy demand and costs (System 5 BlaD)
(initial parameter marked in the figure)

Toilet costs are one of the system components for which values are difficult to determine; these costs are affected by economies of scale and learning curve effects that can be expected with regard to prices of innovative toilets. Therefore, a parameter variation is carried out to study the effect of decreasing toilet prices. The initial results shown in Section 4.2 include rather conservative estimates for toilet price averages, based on available references. However, cheaper products are becoming available, and future price decreases can be expected if source-separating systems are going to be implemented on a large scale. Figure 4.30 illustrates the changes in total costs of the greenfield setup dependent on the price per toilet[98]. The initial toilet prices and the resulting total costs are marked in the figure. It can be seen that the relative cost advantages of Systems 3 and 6 as compared to the current situation, can be further increased if costs of urine diverting toilets decrease. However, the total costs of Systems 4 and 5, which are based on vacuum sanitation, do not become less than the costs of the other systems even at toilet prices less than 200 €.

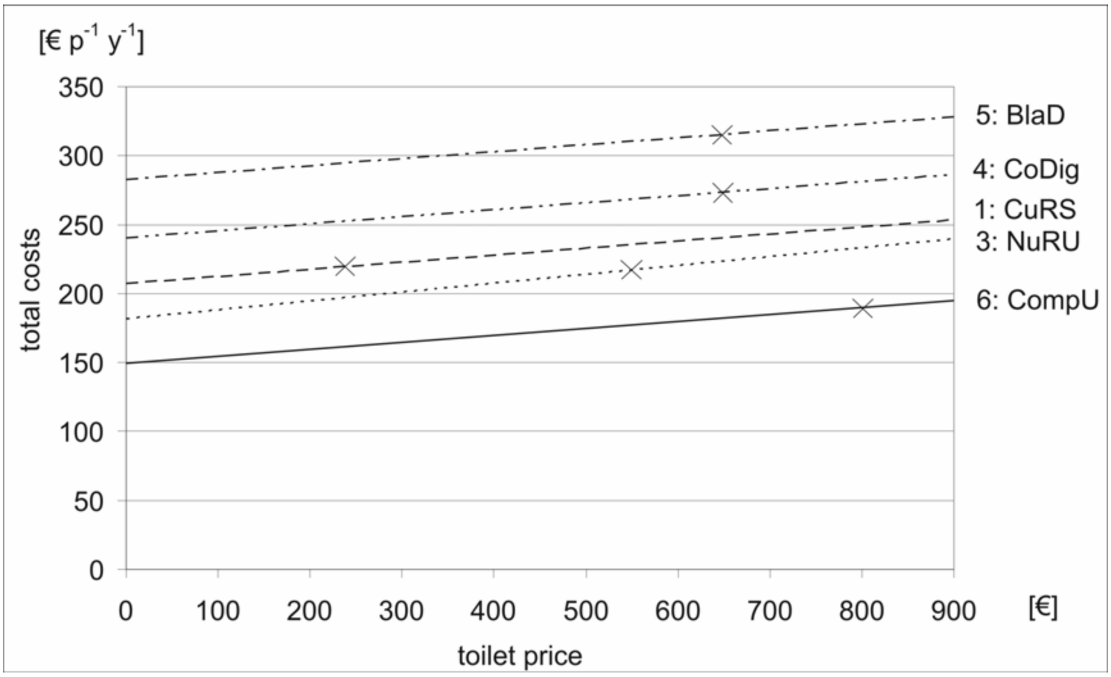

Figure 4.30: Impact of varying toilet prices on total costs (Systems 1, 3, 4, 5, 6 – Greenfield) (initial values marked in the figure)

As discussed in section 2.4.1, prices of phosphorus fertiliser are expected to increase in future due to the finiteness of phosphate rock reserves. In order to evaluate the impact of increasing phosphorus prices on potential benefits of the systems, a parameter variation is carried out which is shown in Figure 4.31. Depending on the ratio of P recovery to the recovery of other nutrients, the specific benefits of the different systems increase with a varying gradient. A comparison of the resulting benefits with the costs for recovery processes[99] shows that System 2 NuRS becomes profitable at a phosphorus

[98] Please note that system 2 NuRS is not included, since this system includes conventional toilets and the total cost changes are analogue to system 1 CurS.

[99] Only sheer costs for recovery processes are included, but not additional costs related to alterations of the overall system, i.e. additional piping or source-separating toilets. For a list of these recovery costs please refer to Table 4.10.

price of about 7450 € t_P^{-1}, System 6 CompU at a price of 7500 € t_P^{-1} and System 4 CoDig at a price of 11350 € t_P^{-1}. The benefits of System 3 NuRu already exceed the costs of stripping and precipitation. By contrast, even rising phosphorus prices cannot generate benefits that are large enough to offset the costs of the recovery processes of System 5 BlaD.

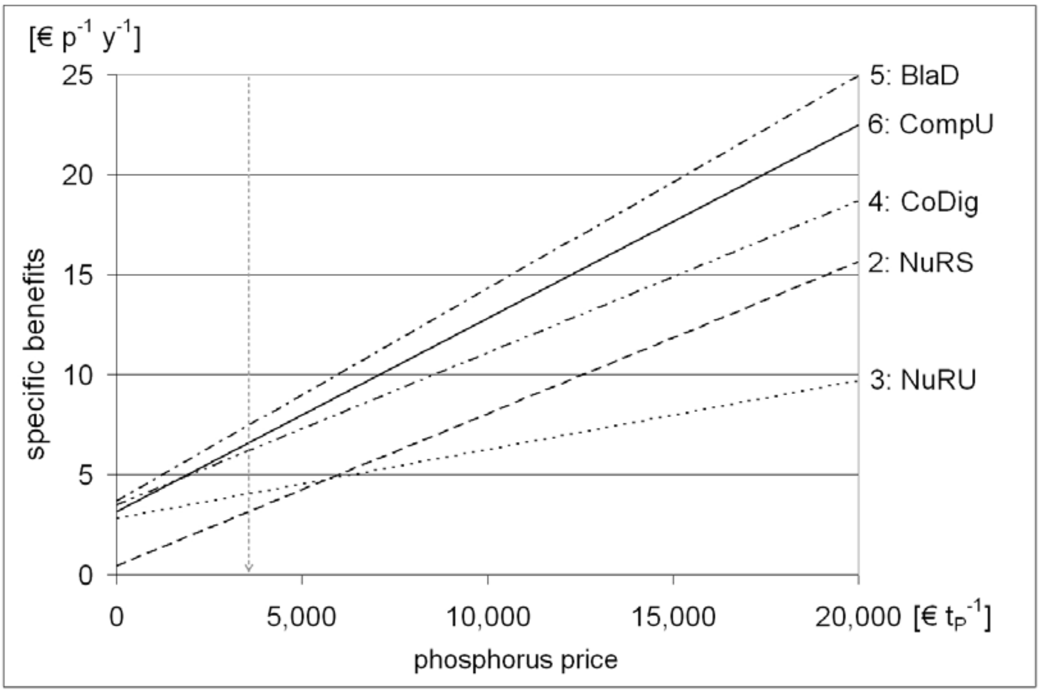

Figure 4.31: Impact of varying prices of phosphorus on benefits from nutrient recovery (initial parameter marked in the figure)

4.3.3 Modifications of the systems

The selected systems represent only an extract of possible system variations. Processes included in the systems can be modified in order to check the possibilities for improving the systems' performances. This goes beyond the variation of specific parameters as discussed in the previous section, and refers to changes in the general system setup and in specific processes. In what follows, some possible modifications for each system are suggested (see Table 4.12); the impact of such modifications on the results is also assessed.

Table 4.12: Measures included in the system modifications

System	Modification
1 CurS	Reduction of toilet flush water
2 NurS	AshDec as P recovery process
3 NuRU	Improvements in nutrient recovery processes (stripping and MAP precipitation)
4 CoDig	Replacement of vacuum system Thickening of digested slurry Omission of N recovery (stripping)
5 BlaD	Phasing out of bottled water
6 CompU	Heat recovery

1 CurS

Current developments show a trend towards a decrease in demand for toilet flush water. The volume of cistern flush can be reduced by do-it-yourself kits or new toilet models can be installed. For example, a low flush toilet with dual flush (2 l/4 l) uses about 14 l p^{-1} d^{-1} (af Petersen et al., 2001). Therefore, the modelling of System 1 CurS is carried out with this reduced flush water volume instead of the previous calculation with 30 l p^{-1} d^{-1}. As a result, the required drinking water and the wastewater inflow to the WWTP are reduced by about 15% to 16%. This also cuts energy requirements and costs for water supply (minus 15% and minus 2% respectively). Also those processes in wastewater disposal and treatment that are flow-dependent, such as pumping, show decreased energy demand and costs (minus 10% and minus 1% respectively). The effort required for nutrient elimination in the WWTP increases due to the fact that the effluent needs to comply with standards for concentrations, which in turn are negatively affected by a decreased water volume and constant nutrient loads. For example, energy and cost requirements for nitrogen elimination increase by about 2%, but the overall cost and energy balances of the WWTP decrease slightly by about 1-3%.

For the overall cost calculation of this modification, a toilet cost of about 360 € is assumed opposed to 275 € for a conventional toilet. This outweighs any potential savings in water supply and wastewater treatment and results in a total cost increase of 1% for the whole system.

2 NuRS

The technology underlying the nutrient recovery process from sewage sludge in the model is the Seaborne® process. This process is selected due to data availability and the existing pilot plant. Other technologies are currently under development (see also Section 2.4.3) and are also tested in large-scale. Their practicability and success will need to be assessed in the future. One of the more promising technologies is the so-called AshDec process, developed within the scope of the EU 6th Framework Project SUSAN (see also http://www.susan.bam.de/). Sewage sludge ash from mono-incineration is subjected to several thermo-chemical processes resulting in a licensed fertiliser product called PhosKraft® (Hermann, 2008). For detailed modelling not enough data is available yet. However, some approximate calculations can be carried out using the basic information given in Annex A.8.

Overall, the results (see Table 4.13) are roughly in the same range as for the Seaborne® process, but general improvements in terms of recovery potential, energy consumptions and costs can be observed. In addition to the evaluation shown here, other factors such as ease of implementation, susceptibility to failure, simplicity of operation, etc. will be decisive as to whether one or another specific technology will become widely accepted.

Table 4.13: Comparison of AshDec and Seaborne results

	Value		Unit
	AshDec (i.e. PhosKraft®)	Seaborne®	
Recycled P	1,080	760	$t_P\ y^{-1}$
Recycled N	-	1,850	$t_N\ y^{-1}$
Recycled K	3,400[a]	-	$t_K\ y^{-1}$
Mass flow fertiliser	~20,000	~17,000	$t\ y^{-1}$
Costs	2.0-3.1	6.1	€ $p^{-1}\ y^{-1}$
Energy consumption	6.2-9.2	11.4[b]	kWh $p^{-1}\ y^{-1}$

a) Potassium is added to the process as potassium chloride or potassium sulphate and therefore the recycled potassium is not waste- or wastewater-borne
b) Net energy consumption (i.e. minus savings in WWTP)
- not applicable

3 NuRU

As mentioned in Section 4.2.5 treatment of urine is only more energy-efficient than the transport of untreated urine if the distance between intermediate storage and agriculture exceeds 110 km. However, treatment (i.e. MAP precipitation and stripping) has additional advantages such as the reduction of micropollutants, easier application and last but not least the acceptance of the products by farmers. The overall efficiency of the system could be improved by measures to reduce the energy consumption of the recovery processes. For example, heat recovery from the existing sludge treatment at the wastewater treatment plant or from thermal waste treatment, could reduce the heat consumption of the stripping process. Energy savings for the stripping process of up to 40% seem realistic with improved process setups (Tettenborn et al., 2007). Regarding the overall energy consumption of the system this would decrease the specific energy demand by about 2% to 544 kWh p^{-1} y^{-1}. Regarding MAP precipitation, the cost of magnesium is one of the main contributors, making up about 65% of the total costs (Esemen and Dockhorn, 2009). Esemen and Dockhorn (2009) show that the use of seawater containing high levels of magnesium, reduces the cost by about 75% compared to conventional operational supplements. Other substrates rich in magnesium, such as the wastewater from potassium mining, could also possibly be used for precipitation. Considering the overall costs of the system, this cost reduction seems negligible (less than 0.5%). For the detailed design of the processes however, such saving potentials should be considered.

4 CoDig

Besides the reduction of flush water and the addition of more organic waste (see also Figure 4.28 and Figure 4.29), other measures are feasible to reduce the energy consumption of System 4 CoDig. For example, the vacuum system (toilets and sewerage) for the collection of blackwater could be replaced by extreme-low-flush toilets that use as little water for flushing as vacuum toilets (e.g. 0.6-1 l per flush) but have a lower energy consumption (af Petersen et al., 2001). If the energy consumption for the vacuum sewerage is disregarded, the total energy consumption is reduced by about 16%. However, the flushing ability and other practical experiences need to be considered in more detail to come to a decision as to which of the two approaches is more favourable. Another innovation worthwhile to be mentioned is a patent hold by Hamburg Wasser (Li, 2007). This patent is for a vacuum blackwater collection system connected to a centralised vacuum source that theoretically reduces the energy consumption when compared to conventional vacuum systems. However, no practical experiences have been reported yet, and thus no data on possible energy savings are available.

Furthermore, thickening of the collected blackwater before transport to the centralised treatment is an option that could possibly cut down on energy demand. Assuming the installation of decentralised thickening devices, such as gravity thickeners, the total solids content of the blackwater could be increased about threefold. This would reduce the volume of the blackwater to about one third of the original volume. The excess water (sludge liquor) can be discharged into the centralised sewer system, together with greywater. Assuming an energy demand for thickening of about 10 Wh $m^{-3}_{blackwater}$ and an annual cost of approximately 0.74 € y^{-1} $m^{-3}_{blackwater}$ (all data based on Puchajda and Oleszkiewicz, 2008), this measure would require less than 1 kWh p^{-1} y^{-1} and 2 € p^{-1} y^{-1}. Potential benefits outweigh these requirements due to reduced transport requirements. Energy savings could amount to about 13 kWh p^{-1} y^{-1} and cost savings could be in the range of about 18 € p^{-1} y^{-1}. Therefore, the total energy demand could be decreased by about 2% and total costs could be decreased by about 6%. Thickening of blackwater could possibly also increase the biogas yield due to longer retention times. Puchajda and Oleszkiewicz (2008) report that about 27% more energy can be produced by digestion of thickened sludge with a TS of 6% instead of 3%.

The nitrogen recovery in System 4 CoDig, which is based on stripping of nitrogen from sludge liquor, eliminates the need for nitrogen removal in the activated sludge reactor. Also energy from fertiliser production is saved. However, an assessment of a modified system, that does not include any nitrogen recovery, reveals that the overall energy balance and the overall cost balance could both be decreased by about 3% if the stripping process is omitted.

5 BlaD

System 5 BlaD achieves the highest recovery ratio, yet it is the most unfavourable system with regard to energy demand and costs. This is also partly due to the assumption that bottled water is used for drinking and cooking purposes. A better choice would therefore be to treat the recycled greywater to a level where the water is fit for human consumption. However, social perception and acceptance would need to be ensured[100]. Another option for this system is a centralised water supply as in Systems 1 to 4. The locally treated greywater could then be infiltrated, discharged into the nearest watercourse or used for purposes such as irrigation. The resulting energy savings amount to about 20% of the total energy demand, decreasing the specific demand to about 709 kWh p^{-1} y^{-1}. However, this value is still greater than the energy demand of the other systems due to the relatively high energy consumption of the vacuum system and the anaerobic digestion under current assumptions.

Only a combination of measures discussed in Section 4.3.2 (i.e. addition of more organic waste, reduction of flush water) and an alternative provision of drinking water instead of bottled water, could bring down the energy consumption to values comparable to the other systems. For example, a centralised drinking water supply, the addition of 80%

[100] In future, additional sources of drinking water might be developed including decentralised water supply options. For example, Hristovski et al. (2009) suggest that water generation from household energy production by hydrogen fuel cells would be sufficient for potable water supply for human consumption.

organic waste to digestion, and a reduction in flush water consumption down to 4 l p^{-1} d^{-1} would reduce the energy consumption significantly to 597 kWh p^{-1} y^{-1}; a reduction of about 33%. This value is even lower than the current energy consumption, highlighting the potential for possible energy savings. Also the replacement of the vacuum system by a low-flush toilet system as discussed above, could be a measure to reduce the energy demand. Replacing bottled water by a centralised water supply alone, could decrease costs by about 22% to 267 € p^{-1} y^{-1}. This is still greater than the costs of the other systems, but in a comparable range.

6 CompU

System 6 CompU could be adapted by the introduction of urine treatment or the replacement of recycled water by a centralised water supply (as discussed for Systems 3 NuRU and 5 BlaD). The expected improvements in terms of energy and costs are considered to be rather marginal, so that these measures are not discussed here in detail.

Another measure that could be an appropriate improvement, not only for System 6, but also for the other systems, is heat recovery from wastewater or greywater. This measure is worthwhile to look at, although energy requirements for household purposes such as heating of water for showering, etc., are not included within the system boundaries of this study[101]. Heat recovery from wastewater (i.e. connected to the sewerage system), which could be an addition to Systems 1 to 4, is currently being tested by Hamburg Wasser (Werner and Augustin, 2009). Heat exchangers can be used to utilise the heat in the wastewater for heating or cooling purposes. For more general information on this topic please refer also to Section 2.5.2 and BFE (2009). Also, the decentralised heat recovery from greywater[102] directly by local energy recovery, seems to be a promising alternative, e.g. by recovering the heat from spent shower water for heating water. Cooling 1m^3 of greywater by 1°C can theoretically provide 1.16 kWh, but there will be losses in heat exchanger and heat pump devices. The coefficient of performance of a heat pump fed with shower water can be up to about 10 (Menerga, 2009). According to Forstner (2009) about 15 kWh can be recovered from 1000 l greywater. Taking the average water consumption for showering into consideration (about 30 l $p^{-1}d^{-1}$),

[101] Another possibility to save energy, that shall be briefly mentioned here, are appliances to reduce the consumption of warm water, for example, water-saving shower heads (anti-legionella devices should be preferred).

[102] Greywater has on average a higher temperature (about 28°C to 40°C) than mixed wastewater (about 12°C to 15°C) since it originates from activities such as showering and washing with warmer water than, for example, toilet flushing.

164 kWh p^{-1} y^{-1} could be recovered. This is about 26% of the required energy demand in System 6, which, however, doesn't include any energy for heating or cooling. Assuming a cost[103] of about 400 € p^{-1}, a lifetime of 12.5 years and an interest rate of 3%, the annuity of this measure amounts to about 39 € p^{-1} y^{-1}, which is an extra charge of about 18% of the total specific costs of System 6.

4.4 Transformation processes

Up to now the system analysis focussed on rather tangible criteria. In addition, decision-making processes need to be seen as embedded within a framework of other criteria and motivations. This section therefore gives a brief overview of the broader picture, including external transformations, possible barriers and potential starting points. The discussions are valid not only for Hamburg, but refer also to the general situation in Germany. Despite this, reference is made to local specifics for the case study Hamburg, wherever this is deemed relevant.

4.4.1 Drivers for change

The main focus of this thesis is on the assessment of specific criteria related to efficient resource use. However, changes in the current water and wastewater system can occur due to a variety of drivers and evolutionary (economic and social) transformation processes. Therefore, this section aims at assessing the described systems regarding their behaviour related to such possible changes in the economic, environmental and social framework. Several authors have discussed the current transformation processes and the need for a paradigm change in wastewater management in Germany (see, for example, Herbst, 2008; Koziol et al., 2006; Libbe, 2007; Riße and Herbst, 2004; Schaller et al., 2007). A task group on new sanitation concepts of the DWA (German Water Association), carried out a Delphi study assessing the importance of certain criteria acting as drivers for, or barriers to the implementation of new sanitation concepts (DWA (ed.), 2008). The study shows that the drivers are anticipated to change over time and that it is expected that several criteria such as resource recovery and resource use will gain importance over the next 20 years. Considering the systems investigated in

[103] According to supplier information the costs for such a system decrease with number of connected showers. With three connected showers the specific investment cost is about 580 € p^{-1} and with 20 connected showers it is about 370 € p^{-1} (Haase, I., 2009, Kosten ThermoCycle WRG, pers.comm.).

this study, the following factors might act as drivers for the transformation[104] of the current water and wastewater system in Germany:

- Emissions: Emissions to surface waters and groundwater, e.g. via combined sewer overflows or WWTP effluents, need to be further reduced to improve water quality as postulated by the European Water Framework Directive. Hamburg has already invested about 800 million € to reduce the pollution from combined sewer overflows (HSE, 2004).

- Natural water cycles: Global climate change has an impact on the German water system. On the one hand, the increased occurrence of heavy rains together with increased sealing of surface areas aggravates the risk of flooding. On the other hand, dry periods can lead to pressure on natural water resources in certain regions. In Hamburg, every year more than 130 ha of land is developed for settlement and infrastructure, increasing rainwater runoff. At the same time, more and more intense rainfalls are observed leading to flooding of streets and houses (Kopp, 2007). Decentralised systems for rainwater (and greywater) management present a chance to counter these challenges. Authorities in Hamburg have therefore already started to emphasise the need for a shift in rainwater management (BSU, 2006).

- Limited phosphorus reserves: As discussed in Section 2.4.1 there are major concerns regarding the availability of phosphate rock reserves that are economically viable for extraction. With increasing scarcity prices will increase; alternative sources of phosphorus must therefore be developed.

- Finite fossil fuel reserves: Prices on the energy market are expected to increase due to increased demand and increased fossil fuel scarcity. In addition to the further exploitation of renewable energy sources, energy saving and more efficient use of energy become mandatory.

- Micropollutants: Micropollutants, e.g. from pharmaceuticals and personal care products, are entering wastewater and pose a risk to human health as they are contaminating natural water resources. They are evermore becoming a concern and innovative strategies for either eliminating them at source or reducing them at the end of the pipe are needed.

[104] It s interesting to note that Libbe (2006) distinguishes between adaptation processes with moderate modernisations on the one hand and transformation processes on the other hand that are always connected with disruptions and transitions comprising a high degree of uncertainty.

- Demographic change: Demographic change is related to shrinking cities in many regions of Germany (in only a few regions, like Hamburg, are populations growing), suburbanisation processes and an aging population. But also behavioural change, e.g. decreasing water consumption, can jeopardize the operability of the current system. For example, per capita water consumption in Hamburg is expected to decline to about 100 l p^{-1} d^{-1} in the next 20 years (Maass, 2008). Infrastructure systems that are flexible to react to these changes show advantages compared to systems designed for static conditions and with long depreciation periods.

- Reinvestments and rehabilitation: Many facilities of the German wastewater infrastructure have reached the end of their economic lifetime. Therefore, per capita expenditures for reinvestments are expected to increase in the coming decades. Particularly sewer rehabilitations are going to present high costs for the German water sector[105]. Capital-intensive (re)investments such as large sewer systems or treatment plants always represent high sunk costs. A positive response to system transformation opens up opportunities for long-term cost-savings.

- Export of expertise and equipment: The water and sanitation sector in Germany could represent a lead market if innovations are taken up and specific requirements for export are integrated into the design of the systems. Experience with more decentralised and more flexible systems is expected to result in greater export possibilities. Hamburg Wasser is already involved in the implementation of innovative sanitation concepts in countries such as China and India. There are many regions worldwide that are in need of upgrading their sanitation systems, e.g. Eastern Europe, South America, Africa.

The investigated systems are evaluated against the above-mentioned drivers (see Table 4.14). The scoring indicates to which degree one particular driver is favourable to the respective systems. The indicators according to which scores are allocated, are shown in the table.

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[105] In addition, private sewers are going to require immense re-investment in the near future. Thoma (2006) estimates that about 1250 to 2500 € p^{-1} of private investments will be required for inspection and renovation in the coming 20 to 40 years.

Table 4.14: Evaluation of the systems regarding possible drivers for transformation

(0 = system shows no advantage, + = system is beneficial, ++ = system is very beneficial)

	2 NuRS	3 NuRU	4 CoDig	5 BlaD	6 CompU	*Indicator*
Emissions	0	+	+	++	++	*N&P emissions to surface waters, see Table 4.5 & Table 4.6*
Natural water cycles	0	+	+	++	++	*Groundwater extraction, see Figure 4.14, & decentralised rainwater management*
Phosphorus	++	+	++	++	++	*Recovered P, see Figure 4.12*
Fossil fuel	0	++	0[a)]	0[a)]	0	*Energy consumption, see Figure 4.15*
Micropollutants[b)]	0	+	++	0	0	*Separation of urine & faeces from wastewater, use of treated products*
Demographic change	0	0	+	++	++	*Higher degree of decentralisation, more flexibility*
Reinvestments[c)]	0	0	+	+	+	*Decentralised facilities, omission of sewer renovation*
Export	0	+	+	++	++	*Low water use, innovative & flexible technology*

a) This system can be adapted to become more energy efficient as described in Section 4.3.3
b) Particularly micropollutants in urine and faeces such as pharmaceuticals are considered
c) From a pure cost perspective none of the systems is more advantageous than the current system

4.4.2 Preconditions

The implementation of alternative[106], more resource efficient water and wastewater systems depends greatly on committed policy entrepreneurs or "enthusiasts" with a long-term vision (Tidaker, 2007). They will aim to create political support on a local and regional level to start transformation processes. Public law might need to be adapted (especially regarding reuse) and particularly for pilot installations financial support needs to be provided. Tidaker (2007) mentions that a shared perspective and a coordinated approach are important prerequisites for successful system change. For this, it is essential to involve all stakeholders already at an early phase to initiate the

[106] The focus of the following discussions is particularly on systems requiring a more radical change such as systems 3 to 6, and not on purely technical adaptations as in system 2 NuRS.

transformation process. In the case of changing the current wastewater system, this includes not only municipalities, water authorities and water utilities, but also real estate developers, housing associations, inhabitants, architects, craftsmen and manufacturers. In addition, it is indispensable to involve agricultural stakeholders from the very beginning if wastewater products are expected to replace mineral fertiliser and to be integrated into farming activities. For farmers to become actively involved, sufficiently large volumes might be needed to make their efforts worthwhile. Therefore, projects need to be large enough to generate sufficient recovery volumes. As has been shown in the analysis of nutrient recovery potential, mineral fertiliser can never be fully replaced. Therefore, a thorough understanding of what can and what cannot be achieved by alternative systems is important.

4.4.3 Starting points

The transformation of the current sanitation system cannot be done on a short-term basis, but requires successive changes in the medium- or long-term. Suggested starting points would be areas that show problems under the current system and where one of the drivers as discussed before, applies. For example, in areas where demographic changes challenge the existing infrastructure or where heavy rains result in frequent flooding, political will and social acceptance for a system change might be relatively easily established. The economic analysis in Section 4.2.6 showed that the implementation of source-separating systems can be done at a comparatively lower price in new buildings or greenfield developments, than in the case of existing housing stock. The same applies to modernisations and refurbishments. Therefore, housing cooperatives[107] that gradually modernise their housing stock, are a useful partner for finding possible starting points. Public places or buildings can be another starting point, where source-separating systems can be implemented. Particularly the collection of undiluted urine from waterless public toilets represents a straightforward possibility for resource recovery. Systems with decentralised structures can represent an alternative in areas that already have a decentralised infrastructure like rural areas, or in new development areas at the fringes of larger towns. The type of urban form can also have an impact on the cost of the respective systems, since criteria such as population density, housing layout and housing types have a direct effect on for instance, pipe requirements (Meinzinger et al., 2010).

Of great importance is the identification of windows of opportunity. Infrastructure in the water and wastewater sector is usually characterised by high investments and long

[107] In Hamburg, more than 29% of the population lives in housing stock owned by housing cooperatives (source: http://www.hamburgerwohnline.de and http://www.saga-gwg.de, last accessed: 20 Feb 2010).

lifespans. Sewer systems in Germany usually have a lifespan of about 80-100 years. Also, the structural part of wastewater treatment plants is projected for a use of about 50 years. That means recovery periods for long-lasting infrastructure need to be taken into consideration and renovation requirements demanding re-investments need to be identified. In these areas decisions for a system change can be made without questioning the economic viability of the whole system. In addition, urban renewal and urban transformation processes can represent suitable starting points.

Another factor to consider is the possible improvement of the overall performance and efficiency of the existing wastewater management. For example, as shown in the analysis of the systems, source separation can have an effect on the energy balance of the wastewater treatment plant. Also the capacity of existing facilities can be increased by the introduction of source separation.

4.4.4 Challenges

Since a system change does not take place at once, the integration of new approaches to sanitation with existing infrastructure seems very important. Interactions between new and old systems need to be considered. For example, although considered to be advantageous for the energy balance of the wastewater treatment plant, the separation of blackwater can impact on the volume of wastewater in the sewer to an extent where difficulties arise in sewer operation. These considerations will need to be taken into account on a case by case study.

In addition, a system change not only involves technological changes, but can also affect organisational issues (i.e. decentralised facilities operated by a centralised utility) and of course social aspects. Public acceptance for novel sanitation facilities (e.g. source-separating toilets) is crucial. Even though pilot projects up to now have been well received by users[108], large scale implementations will require a broader public acceptance, in particular towards reusing human waste products. An end-user perspective, including the winning of trust will be important for successful transformations.

A system change will generate questions about who is going to cover the costs, particularly if costs are transferred from centralised utilities to households for example, as shown in Section 4.2.6. Additional costs, which might occur particularly in the

[108] It should be noted here, that in Germany public perception towards dry toilets (as in System 6 CompU) is rather biased. Pilot implementations in ecologically oriented housing projects exist, yet, this technology is at the current status of development not considered to be an option for large-scale implementation. However, improvements to arrive at a broader level of acceptance are imaginable.

beginning, when economies of scale do not yet apply, need to be covered, keeping in mind the benefits that can be created in the long run. As human waste products are not yet monetarily valued according to their fertilising potential, it will be important to work further into the direction of marketing recycled products. More research needs to be done to eliminate any uncertainties regarding nutrient values and the safety of products. Yet, it should also be considered that at current financial values of human waste products, recycling might not be competitive from a purely economic perspective, due to the high costs involved in recycling processes. However, because of the variety of transformation processes and current trends in price developments, the sanitation sector should be prepared for a paradigm change.

Adaptation and transition periods need to be used to find systems that are capable of yielding benefits in terms of resource recovery, cost saving and energy saving, without compromising the efficiency of the whole system[109]. Socio-economic transformations and other external factors (as discussed in Section 4.4.1) need to be integrated into the long-term infrastructure planning strategies. Further experiences and thorough assessments of pilot installations are fundamental for decision making and system transformations.

Another interesting question is how and by whom a system change should be introduced. Most often it is implied that adaptations in infrastructure are regulated by governmental bodies or large-scale infrastructure providers. However, particularly in the case of more decentralised facilities, change can also be initiated by end-users[110]. For example, technologies such as greywater recycling units for private installations could be offered for sale in hardware stores. Such changes at the source on a larger scale will necessarily require utilities and service providers to adapt accordingly. Strategic planning is therefore inevitable and should consider the needs and perspectives of all stakeholders.

A further discussion of planning and decision making in consideration of increased resource efficiency can be found in the concluding discussion in Section 6.2.2.

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[109] Hegger (2007, p.193) calls them "modernised mixtures" and describes them as "late-modern socio-technical configurations (of wastewater infrastructures) in which various features of simple modern systems have been deliberately and reflexively reconstructed to deal with contemporary social, economic and environmental challenges."

[110] For example, a comparable development can be noted in the energy sector where more and more decentralised units such as combined heat and power plants or solar collectors are installed.

5 Arba Minch

Analogue to the presentation of the ceMFA for the case study Hamburg in Chapter 4, this chapter starts with process descriptions used for the modelling of the Arba Minch case study (Section 5.1). The illustration of the modelling results in Section 5.2 is complemented by a sensitivity analysis and parameter variations (Section 5.3). The chapter concludes with a discussion of possible transformation processes and consideration that are important in the context of Arba Minch.

5.1 Process descriptions

In the following sections the processes and the parameters that are used to calculate the mass and nutrient flows, energy balances and costs are described in detail. For an overview of the processes and flows included please refer to Figure 3.11.

5.1.1 Agriculture

Mass and nutrient flows

Only few stakeholders are active in the farming sector in Arba Minch. One of the main agricultural players is the Arba Minch state farm with a cultivated area of about 820 ha (Plückers, 2009). All in all there is cropland of about 1000 ha within Arba Minch town boundaries (based on Plückers, 2009). Similar to the system analysis for the Hamburg case, the model includes an agricultural area that extends to an area large enough to produce the food for the population of Arba Minch, i.e. including a hinterland area. According to the Ethiopian Food Balance Sheet (FAO, 2009b) the average food consumption per capita in 2003 was about 290 kg p^{-1} y^{-1}, of which the largest share was produced domestically. Since the crop yield in Ethiopia is lower than the yield in Europe, the average area needed to produce one kilogram of food is higher than the 4 m^2 for Europe, as given by Baccini and Brunner (1991) (see also Section 4.2.1). A comparison of the average yields of cereals, roots and vegetables in Germany and Ethiopia based on FAO (2009c), shows that the yields per hectare are at least 3.5 to 5 times higher in Germany than in Ethiopia. Therefore, the average size of cropland needed to produce one kilogram of food in Ethiopia is estimated to be about 15±2 m^2. With a population of about 80,000 people (based on AMU and ARB, 2007; CSA, 2006) and about 12% imported food this translates to an agricultural area requirement of about 30,600 ha in total.

Agriculture in Arba Minch is mainly rain-fed. Only a few sites close to the Kulfo river are additionally irrigated by river water. The parameters that are used to calculate the water flows of the Process Agriculture are summarised in Annex F.1.

Fertiliser application rates in Ethiopia are generally rather low. This is mainly due to financial constraints. As Plückers (2009) revealed, fertiliser application of the main producers in Arba Minch, such as the state farm, is lower than recommended, since rising prices place the use of mineral fertiliser out of reach. In recent years the state farm used annually about 50-80 kg of Urea and 90 kg DAP per hectare[111]. Small scale farmers usually do not apply any fertiliser at all. For the analysis an average annual application rate of 50 kg Urea and 100 kg DAP is assumed, which reflects the use of fertiliser by the Arba Minch University and is in line with the recommendations of the Ethiopia Ministry of Agriculture[112] (Plückers, 2009). This translates to an average required fertilisation with 39.0 kg_N ha^{-1} y^{-1} and 20.1 kg_P ha^{-1} y^{-1} in total. The actual mineral fertiliser inputs to the system are calculated in the model, depending on the amount of nutrients added from other flows such as urine or compost.

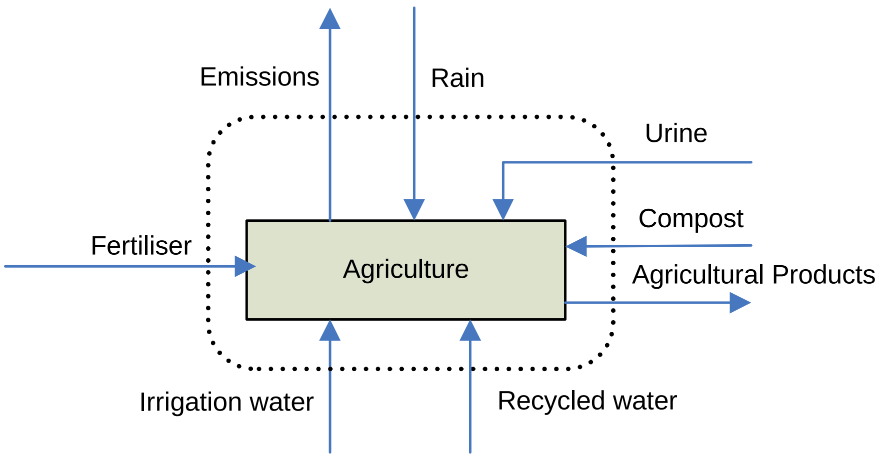

Figure 5.1: Process Agriculture

[111] According to state farm information the recommended rate for their fields would be 100-150 kg_{Urea} ha^{-1} y^{-1} and 200-250 kg_{DAP} ha^{-1} y^{-1} (Plückers, 2009).

[112] It is noticeable that the recommendations for Ethiopia include a relatively higher phosphorus fertilisation and a lower nitrogen fertilisation than in Germany. Yet, it is not clear on which factors these recommendations are based upon.

Energy

There are no energy requirements directly related to the Process Agriculture included in the analysis. However, the same energy requirements for production for mineral fertiliser are used as in the case of Hamburg, which are based on the work of Patyk and Reinhardt (1997) (see Section 4.1.1). Since there is no mineral fertiliser production in Ethiopia and all fertilisers are imported, energy requirements are even higher than these values. To reflect this, the error margin is increased to 20%.

Costs

Costs for the Process Agriculture are neglected in this study. Also the costs for fertilisers are not included, since the input of nutrients from urine and compost is valued as a benefit in the cost analysis. The value of compost is based on the average price paid in Ethiopia, i.e. 1.5±0.2 ETB kg^{-1} (Plückers, 2009)[113]. The value of urine is calculated based on the nutrient flows recycled to agriculture as they are calculated in the ceMFA. The economic benefit associated with the nutrients nitrogen and phosphorus, is calculated according to a method shown by Esemen and Dockhorn (2009) using average mineral fertiliser prices and converting them to nutrient-specific prices by taking into account the respective nutrient content. In Ethiopia, the average price for Urea (46% N) is 5.5±0.5 ETB kg^{-1} and the average price for DAP (46% P_2O_5, 16% N) is 7.0±1.0 ETB kg^{-1} (Plückers, 2009). This can be converted to a benefit of 12±1 ETB kg_N^{-1} and 25±5 ETB kg_P^{-1}.

5.1.2 Households

Data regarding household size and total inhabitants in Arba Minch have a rather high uncertainty, since reporting is error-prone and there is a constant increase due to migration to the town. According to AMU (2007) there are about 80,000 people living in Arba Minch. The average household size is made up of about 5 persons per household (ibid.), which means that there are roughly 16,000 households in Arba Minch. The main input and output flows of the Process Households used in the model are shown in Figure 5.2.

[113] In Arba Minch, the first compost prepared by the Youth Group Egnan New Mayet in 2009 was sold to the municipality for prices as high as 10 ETB kg^{-1} (Plückers, 2009). Yet, this is supposedly due to hidden subsidies and does not reflect the medium term development of the compost prices.

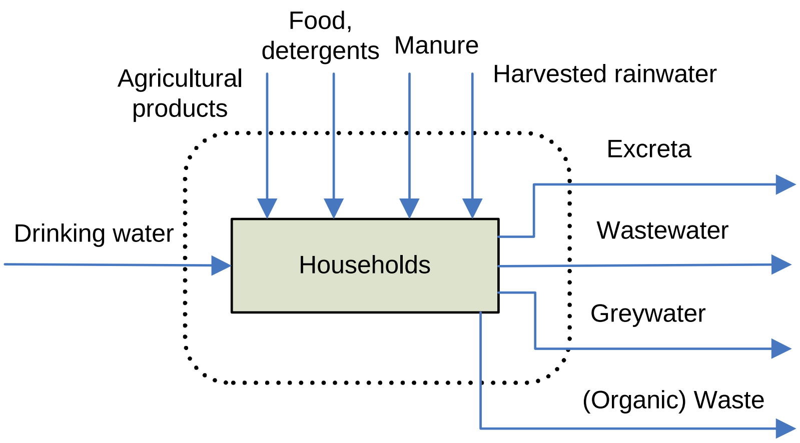

Figure 5.2: Process Households

Mass and nutrient flows

One of the main inputs to the households is water, which is used for various purposes such as drinking, cooking, washing, cleaning, etc. For modelling the water consumption for these purposes is set to be 40±4 l p^{-1} d^{-1} on average for household connections (i.e. yard connection or in-house connection) and 5±0.5 l p^{-1} d^{-1} for those households dependent on public standpipes (see also Section 2.3.1 and Table 5.1). Since currently only a few households in Arba Minch use water-flushed toilets, it is assumed that the above values don't include any consumption for toilet flushing. Therefore, an additional consumption of 6±1 l p^{-1} d^{-1} is included in the model for the households that use flush or pour-flush toilets (particularly in System 4 AnDig). Food is supplied to the households mainly from the market and to some extent from home-grown crops. The mass and nutrient content of the food is derived from FAO data and shown in Annex F.2.

Outputs from the Process Households include organic waste, greywater, urine and faeces. The faeces volume is assumed to be higher than the European average due to a diet higher in fibre and is set at 0.3±0.03 kg p^{-1} d^{-1} (see also Jönsson and Vinneras, 2004). Regarding urine, it can be assumed that the volume is lower than the European average due to increased sweating and lower liquid consumption. It is therefore set at 1.1±0.1 l p^{-1} d^{-1}. The appraisal of nutrient content of excreta can be found in Section 2.4.2. The amount of greywater is calculated according to the water used. Since there is very little data available on nutrient contents in greywater in places like Arba Minch, the nitrogen and phosphorus load in greywater is estimated from Ridderstolpe (2004) to be 0.8 g_N p^{-1} d^{-1} and 0.5 g_P p^{-1} d^{-1}. The amount of organic waste is estimated to be about 0.12 kg p^{-1} d^{-1} (Kuma, 2004). The estimated nutrient content of organic waste is listed in Annex F.2.

Energy

In order to assess the benefit of using biogas to replace firewood and charcoal, the model includes the domestic energy use from these two sources. It is estimated that both energy sources are used in a quantity of 120±24 kg p^{-1} y^{-1} (Abebaw, 2007). The heating value is estimated to be about 4±0.2 kWh kg^{-1} for wood and 8±0.2 kWh kg^{-1} for charcoal (Gloor, 2009). The primary energy demand from fuel woods reduced by the use of biogas is calculated from Equation 5-1:

$$E_{red} = E_{WF} - E_{biogas} \cdot \frac{\eta_{biogas}}{\eta_{woodstove}} \qquad [\text{kWh p}^{-1}\text{ y}^{-1}] \quad (5\text{-}1)$$

with
E_{WF}: current energy demand from wood fuel [kWh p^{-1} y^{-1}]
(current consumption of firewood * heating value of firewood + current consumption of charcoal * heating value of charcoal)
η_{biogas}: efficiency ratio of biogas stove, 0.6±0.05 (Seyoum, 1988) [-]
$\eta_{wood\ stove}$: efficiency ratio of wood stove, 0.2±0.05 (Addison, 2010) [-]

Costs

Costs for firewood and charcoal are not included in the cost analysis, but are used for a separate calculation of benefits. Therefore, the saved expenses for these wood fuels represent the potential benefit of biogas. Cost parameters for firewood and charcoal, as well as for construction and operation of rainwater harvesting, and greywater treatment are listed in Annex F.2.

5.1.3 Water supply

Mass and nutrient flows

Groundwater is the main source of drinking water in Arba Minch. The water is supplied to the households by yard or house connections as well as by public standpipes or water vendors (e.g. neighbours). According to AMU (2007) about 22% of the households buy their water either from public taps or from vendors. The remaining 78% have either a yard connection (67% in total) or a house connection. The assumed water consumption for the respective type of connection is listed in Table 5.1. The water loss in the piping system is indicated to be about 11% (AMU and ARB, 2007), which is considered to be relatively low. In addition, it is assumed that about 5% of the water taken from public taps or vendors is lost. The nitrogen and phosphorus concentrations in drinking water are estimated to be about 1.21±0.10 g_N m^{-3} and 0.11±0.01 g_P m^{-3} (DHV Consultants, 2002). Carbon in drinking water is neglected in this study.

Table 5.1: Estimates of specific water consumption [l p^{-1} d^{-1}]

	house / yard connection		public stand pipe / water vendors	
	average	minimum-maximum	average	minimum-maximum
eating, drinking	5	3-7	3	2-5
washing, laundry, cleaning	35	15-35	2	1-5
toilet flushing	6	0-20	2	0-5

Energy

The required electricity for drinking water supply (i.e. pumping from the source to the reservoir) is calculated according to Equation 5-2[114]:

$$E = \frac{m \cdot g \cdot h}{\eta_{pump} \cdot 3.6} \qquad [\text{kWh y}^{-1}] \qquad (5\text{-}2)$$

with
m: mass (determined from mass flow) [t y^{-1}]
g: gravity [m s^{-2}]
h: pump head (about 350±50m) [m]
η_{pump}: pump efficiency ratio (assumed: 60±10%) [-]

Costs

Water is sold at different prices in Arba Minch. For example, the water sold at public standpipes includes a salary for the attendants. The basic fee set by Arba Minch Town Water Service is 1.5 ETB m^{-3} (AMU and ARB, 2007). In this study it is assumed that this fee roughly represents the cost of the water and it is therefore multiplied by the water consumption to arrive at the cost of water supply.

5.1.4 Wastewater of the university

Mass and nutrient flows

Arba Minch University is a major institution in Arba Minch with a student number of about 8,000 in 2008, and this number is expected to increase to 24,000 by 2014. Arba Minch University has its own water supply, wastewater treatment as well as some farmland. Groundwater extraction from seven wells is estimated to amount about

114 Direct data about actual electricity consumption of the Arba Minch water supply is available (personal communication, Gelaye, B., 27 December 2009, General Manager of Arba Minch Town Water Service). However, the data is variable due to frequent power cuts. The use of suggested formula allows the incorporation of changes in water consumption into the model.

100,000 to 150,000 $m^3\ y^{-1}$ based on the work of Dagelo (2005) and Kröger (2007). This is in line with a per capita water consumption of 37 to 55 $l\ p^{-1}\ d^{-1}$ for the 7,400 students and staff residing on campus. This relatively high value is partly due to water-flushed toilets and a high water loss in the distribution system, which is estimated to be about 21% (Kröger, 2007). The wastewater is treated in two series of waste stabilisation ponds. Part of the pond water is used for irrigation by nearby farmers, which is estimated in the ceMFA model to be 50±20% of the total inflow. The remaining water evaporates, or is discharged as pond effluent to nearby rivers.

Nutrients in the water supply of the university are neglected, but a nutrient and carbon input to the Process University from food is assumed. Excreted nitrogen, phosphorus and carbon are then discharged into wastewater and enter the ponds. Elimination rates in the pond systems are listed in Annex F.4.

Energy

Energy requirements for the water supply of the university are calculated using Formula 5-2 above, but a pump head of only 50±5 meters is assumed. The wastewater treatment of the university, i.e. the pond system, does not require any energy.

Costs

Construction costs for a waste stabilisation pond are calculated according to Equation 5-3 (Loetscher, 2002):

$$C = 3500 \cdot \left(\frac{P \cdot Q}{1000} \right)^{0.74} \qquad \text{[USD]} \qquad (5\text{-}3)$$

with
P: number of persons served
Q: volume of water in litres discharged per person and day

The resulting construction cost is then converted to ETB using exchange rates from OANDA (2009). The lifetime of the pond system is assumed to be 40 years. Operation and maintenance costs are approximated to be 2.5% of capital costs according to Loetscher (2002).

5.1.5 On-site sanitation facilities

Mass and nutrient flows

The Process On-Site Sanitation Facilities represents a principal component of the ceMFA model of Arba Minch; it actually consists of three processes (see Figure 5.3). A differentiation is made between dry toilets (i.e. pit latrines, ventilated improved pit latrines (VIP) and urine diversion dry toilets (UDDT)), anaerobic tanks (i.e. septic tanks and anaerobic digesters) and soil filters for greywater treatment. System parameters are introduced that allocate the respective flows from the households to the sanitation processes depending on the type of system under consideration. This means that parameters are integrated into the ceMFA that indicate, for example, the percentage of households using pit latrines. Also the amount of manure, organic waste or greywater added to sanitation processes is determined by system parameters which are specified in section 3. Greywater treatment in soil filters is an optional process in the ceMFA and is included only in systems 3 and 4. Greywater is also partly added to septic tanks. It is not added to anaerobic digesters in order to allow a high efficiency of the biogas production. The inflows to the sanitation processes, such as excreta, greywater and mixed wastewater, are characterised by the parameters listed in Annex F.2 and F.6.

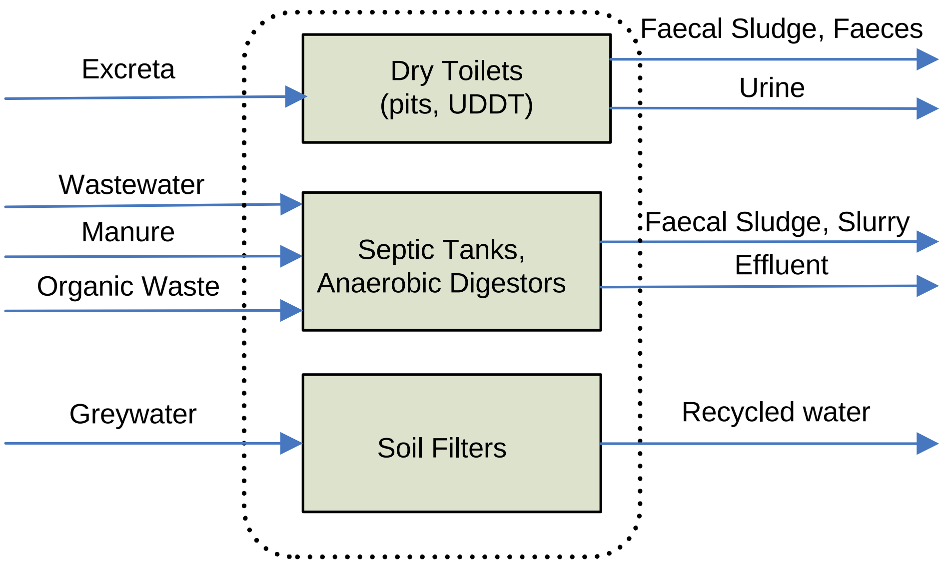

Figure 5.3: Processes On-Site Sanitation Facilities

Mass and nutrients undergo different reactions within the processes, such as ammonia volatilisation or incorporation into particular flows such as faecal sludge. Transfer coefficients as listed in Annex F.5, are selected to describe the different processes as black-box models. The urine separating efficiency of UDDT is assumed to be 70% (Jönsson, 2001). The processes in septic tanks and anaerobic digesters are assumed to be the same, with the difference that there is no use for faecal sludge and gas from septic tanks. The treatment efficiency of soil filters is assumed to be lower than in Europe, since they are less diligently operated and controlled in Ethiopia[115]. In addition, they are

[115] In Arba Minch, so-called greywater towers are in place above ground (Ayele, 2009), these towers are easier to construct but more prone to overloading than conventional subsoil filters.

purpose-built for reuse of the water for crop production; nutrient elimination is therefore actually less important.

Energy

In System 4 AnDig anaerobic digesters are in place, which treat and produce biogas from organic waste, animal manure and blackwater. The energy production is calculated according to the organic matter of the input materials, i.e. the volatile solids content. A biogas production of 0.5 $m^3\ kg_{VS}^{-1}$ and a heating value of the biogas of 22.5 MJ m_{biogas}^{-3} is used in the model (Müller et al., 1999; Thomé-Kozmiensky, 1998). An efficiency of 60±5% for the use of biogas in stoves is assumed (Seyoum, 1988). Further parameters are shown in Annexes F.2 and F.5.

Costs

Cost parameters for the different types of sanitation facilities (investment as well as operation) are listed in detail in Annex F.5. Total costs are calculated by multiplying the respective unit costs with the percentage of households using the specific type of sanitation (as defined in Table 3.3). One toilet per household is assumed.

5.1.6 Markets and livestock

Mass and nutrient flows

In Arba Minch several markets that serve as transfer points for crops and other goods to the households exist. Organic waste having its origin from the markets is assumed in systems 2 CoComp and 3 UDDT to be added to the central composting process. Alternatively, organic waste from the markets is given to livestock as forage; this is illustrated in Figure 5.4. Organic waste from the households is also partly used as forage[116]. Livestock plays an important role in the subsistence of the people in Arba Minch. According to Plückers (2009) there are about 14,000 livestock units in Arba Minch. This number refers only to cows and cattle. Sheep, goats and poultry are neglected in this study. The feed of a livestock unit, which is usually from grazing, is not fully modelled in this study, but only the use of organic waste is included[117]. The manure per livestock unit can be estimated to be about 13 $t\ LU^{-1}\ y^{-1}$ (Boxberger et al.,

[116] In System 1 CuSit it is assumed that about 30% of the organic waste of households is given to cattle and other animals (Kröger, 2007).

[117] As a result, the model adds a negative balance to the process livestock, but this value should be derived from grazing.

1995). A certain ratio of the manure, which is estimated to be about 10%, is used in the households as fuel. The remaining is usually dumped. But in systems 2 CoComp and 4 AnDig manure is also added to the respective treatment processes, i.e. co-composting or anaerobic digestion. Nutrient concentrations and other characteristics of manure as they are assumed in this study are shown in Annex F.6.

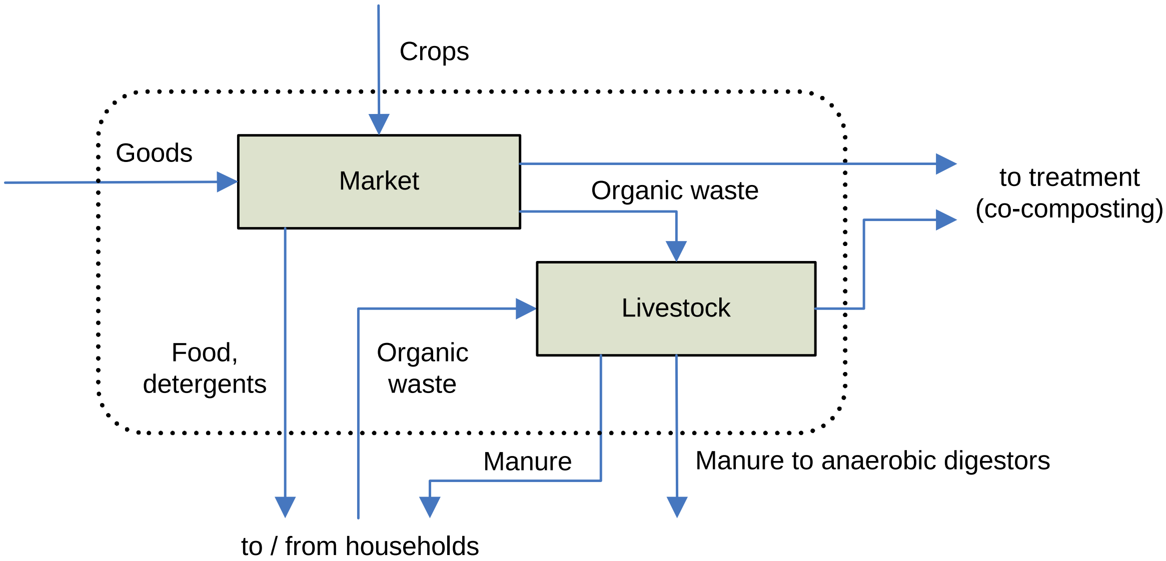

Figure 5.4: Processes Markets and Livestock

Energy

Energy demand or production related to the handling of manure and organic waste is included in the Processes On-Site Sanitation Facilities (i.e. anaerobic digestion) and Transport.

Costs

Costs for the processes and flows related to markets and livestock are not included in the analysis, since it is assumed that the respective costs are constant.

5.1.7 Collection/Transport

Mass and nutrient flows

Vehicle based transport is assumed for transports of waste and wastewater flows in the different systems as listed in Table 5.2. It is assumed that there are no changes in mass and nutrient flows as a result of the Process Collection/Transport. In addition to the collection by vacuum trucks and dump trucks, donkey carts are needed to collect organic waste, faeces, urine and faecal sludge from narrow streets and locations with limited access.

Table 5.2: Lorry-based transports included in the Arba Minch ceMFA

Transport of	Average one-way distance [km]	System
organic waste from households to co-composting	15±5	2 CoComp, 3 UDDT
organic waste from markets to co-composting	5±2	2 CoComp, 3 UDDT, 4 AnDig
faeces from households to co-composting	15±5	3 UDDT
faecal sludge from households to co-composting	15±5	2 CoComp, 4 AnDig
compost to agriculture	5±2	2 CoComp, 3 UDDT, 4 AnDig
urine from households to agriculture	15±5	3 UDDT

Energy

Due to inadequate data of the street system in Arba Minch, the calculation of transport requirements is based on estimates of the average distance and not on a detailed logistic assessment as in the case study of Hamburg. This average distance is the distance between point of origin and destination, as derived from maps and personal observations (see Table 5.2). Transport needs are calculated by the respective mass flow (as calculated by the ceMFA), the assumed capacity of the truck (see Annex F.7) and the average distance. The required energy is then calculated by multiplication of the average distance with the specific fuel consumption, and the energy content of fuel (see Annex F.7). A factor of 1.1 is introduced for conversion to primary energy.

Costs

A detailed calculation of transport costs for Arba Minch was done by Mindachew (2009). From his study a specific cost for lorry-based transport of 15±1 ETB km^{-1} is derived. This includes capital costs, labour, and consumables such as fuel and maintenance costs. The cost (capital and operation costs) of a donkey cart including two donkeys and one operator is estimated to be 14,500±1,500 ETB y^{-1} (Mindachew, 2009). To estimate the number of donkey carts required, it is assumed that one donkey cart can carry 800±200 t y^{-1} (based on Mindachew, 2009). This capacity is compared to the flows intended for recycling (i.e. organic waste, faeces, urine and faecal sludge) as derived from the mass flow analysis. Thus, the total number of required donkey carts can be calculated and multiplied by the cost per cart.

5.1.8 Treatment / Storage

Composting of organic waste and organic matter from human waste (such as faeces or faecal sludge, which is treated in drying beds beforehand) is integrated into Systems 2 CoComp, 3 UDDT and 4 AnDig. In addition, urine is stored in System 3 UDDT before agricultural application to allow further pathogen die-off.

Mass and nutrient flows

Ammonia volatilisation can be expected during urine storage due to untight tanks, and during decanting. In the ceMFA model this is integrated into Process Dry Toilet (UDDT) by the introduction of a parameter representing the nitrogen losses (i.e. 8±1%, based on Montangero and Belevi, 2007).

The co-composting process combines mass flows of human waste (i.e. faeces and faecal sludge), organic waste and cow manure. The recommended volumetric mixing ratio of organic waste, cow manure and dewatered human waste is 2:1:1 (Neves et al., 2009; Strauss et al., 2003). The ceMFA model however, calculates the input of organic waste added to the co-composting process based on variable parameters set as system variables (see Table 3.3). Afterwards, the balances can be checked and parameters can be adjusted if necessary. Faecal sludge is first dewatered before being added to the co-composting process. Volume reduction of faecal sludge by dewatering is 40±10% (Strauss et al., 2003). Parameters for the composting process itself are derived based on data from other studies in developing countries and not from the Hamburg case study, since the setup of the facility and the processes differ. Transfer coefficients for the sludge treatment and composting process are listed in Annex F.8.

Energy

Since there is no forced ventilation used for composting facilities in Ethiopia, the energy requirements for the Process Composting are neglected in this study. It is furthermore assumed that all required mechanical processes such as turning or mixing is done by manual labour.

Costs

The cost of urine storage is assumed to amount to about 1700 ETB m^{-3} according to current market prices for plastic water tanks. The required volume is calculated based on the mass flow of separated collected urine and a storage period of six months. i.e. the storage volume needs to be large enough to store half of the annual urine mass flow. The useful lifespan of the storage tank is estimated to be about 15 years. Operation and maintenance costs are assumed to be 2% of the investment costs. In addition, salaries for two workers for supervision are included in overall costs.

Costs of the co-composting facilities are calculated according to the work done by Mindachew (2009) who linearly interpolated cost data from co-composting plants in Ghana (Steiner et al., 2002) and transferred the data to Ethiopian conditions. Thus, the investment and operation costs are calculated according to Equations 5-4 and 5-5:

$$CC = 144.11 \cdot Q + 211{,}970 \qquad [\text{ETB}] \qquad (5\text{-}4)$$

$$OC = 26.14 \cdot Q + 6{,}889 \qquad [\text{ETB y}^{-1}] \qquad (5\text{-}5)$$

with

CC : capital costs [ETB]

OC : operation and maintenance costs (including labour costs) [ETB y^{-1}]

Q: volume of sludge to be co-composted (derived from mass flow analysis by assuming a density of close to 1000kg m^{-3}) [m^3 y^{-1}]

The capital costs are discounted and converted to annual costs by using a lifespan of 15 years (based on Steiner et al., 2002).

5.2 Results

The following sections give an overview of the most important results of the four modelled Arba Minch systems. The layout of the assessment is analogue to the Hamburg ceMFA model. The uncertainties are derived assuming normal distribution of all parameters.

5.2.1 Nutrient emissions to the environment

Nitrogen and phosphorus emissions show a very similar distribution among the four considered systems (see Figure 5.5 and Figure 5.6). The largest emissions by far are from free ranging animals, which can be reduced by about 22% in System 4 AnDig by the addition of manure to the anaerobic treatment. This system however, reveals a potential pollution hazard by untreated effluents from biogas plants. In Ethiopia, usually only the solid part of the biogas plant slurry is further treated by drying or composting, but the liquid part is most often discharged untreated. Nutrient discharges from open defecation or open disposal of faecal sludge are reduced by the different systems proposed, through the implementation of more sanitation facilities and proper operation, e.g. emptying and collection routines.

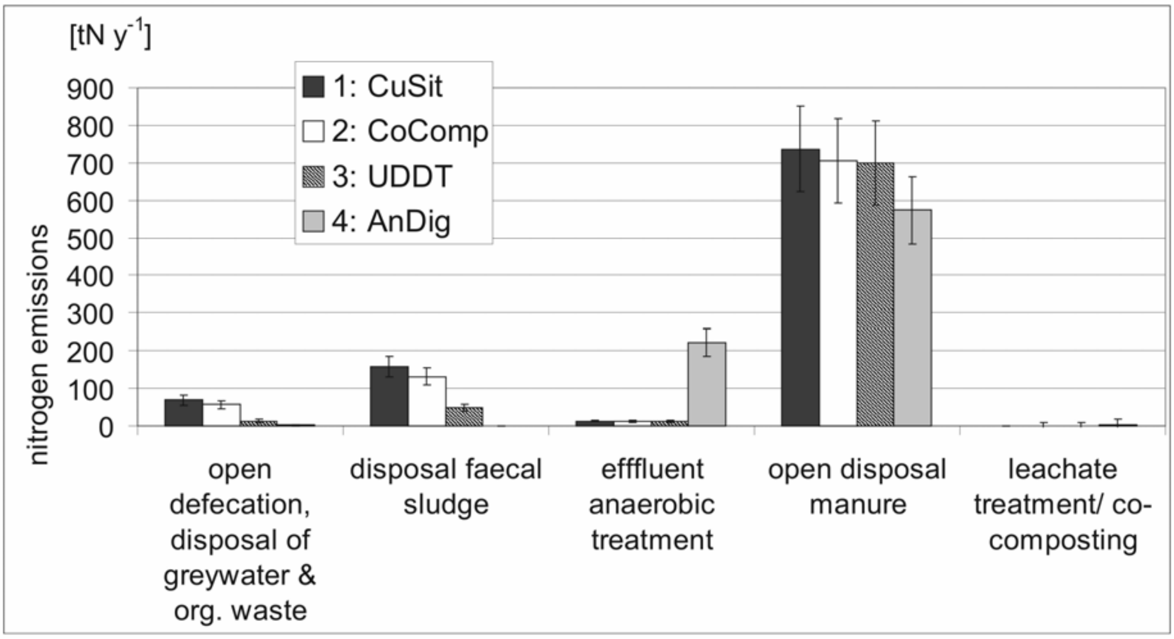

Figure 5.5: Nitrogen emissions to the environment [t_N y^{-1}]

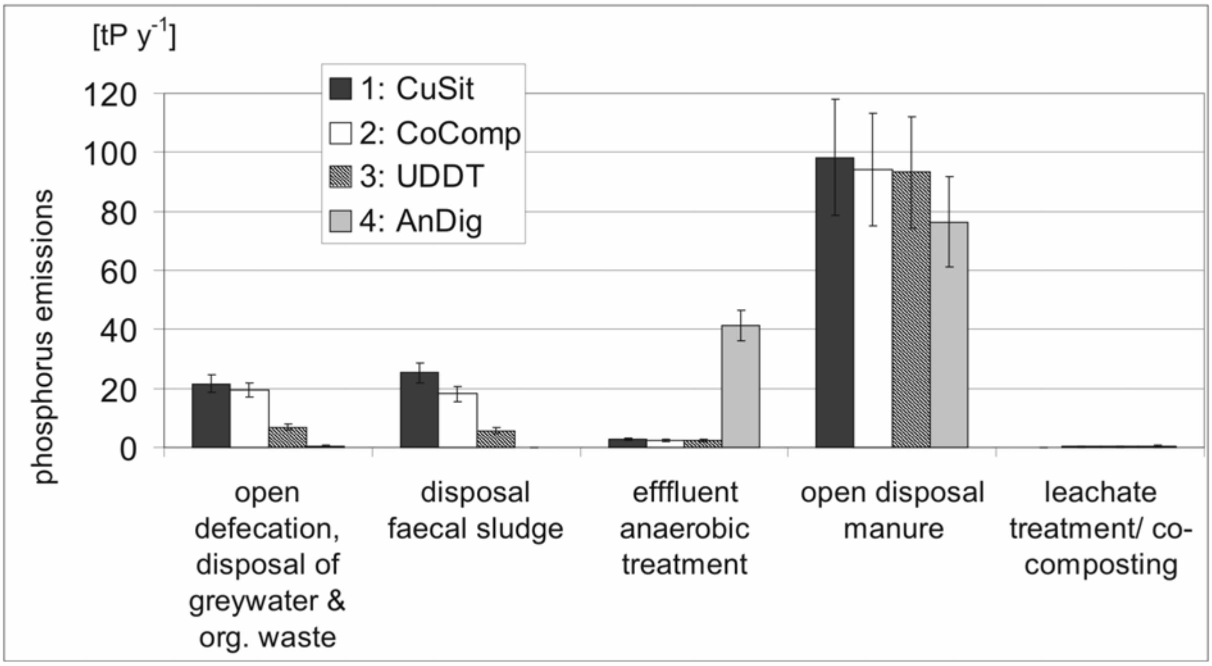

Figure 5.6: Phosphorus emissions to the environment [t_P y^{-1}]

It is important to note that within the scope of this study only a first approximation of expected total emissions to the environment can be given and that no full-risk assessment regarding the pollution hazard is done. For this, more detailed information on topography and hydrogeology of Arba Minch would be needed to assess the risk of nutrients that are discharged onto and into the soil and entering ground- or surface waters. One easy way of reducing the risk of nutrient leaching from pit latrines (i.e. referring to the flow "disposal of faecal sludge"), is to change to a so called arbor loo system. This includes shallow pits that are not emptied when full but planted with trees, which are supposed to take up some of the excreted nutrients. For more information on arbor loos please refer to Morgan (2007).

5.2.2 Recovery of organic matter

In contrast to the Hamburg case study, there are no point emissions of organic matter into surface waters, but only diffuse emissions from the open disposal of waste(water) flows. These are difficult to quantify but show similar characteristics as the nutrient emissions discussed in Section 5.2.1. For the case of Arba Minch however, the recovery of organic matter is a particularly important issue, and it is therefore discussed here. In systems 2 to 4 the process co-composting is introduced, in which different flows such as faecal sludge, faeces or slurry from biogas plants, are treated. As shown in Table 5.3, all three resource-oriented systems achieve a recycling of organic carbon in the form of compost, in the range of 550 to 650 $t_{TOC}\ y^{-1}$. In addition, organic carbon is recovered in the form of methane in biogas in System 4 AnDig, at a rate of about 690 $t_{TOC}\ y^{-1}$. These recovered carbon flows represent valuable soil conditioners for agriculture and potential energy sources supporting resource conservation.

Table 5.3: Recovered organic carbon in compost and biogas [$t_{TOC}\ y^{-1}$]

	1 CuSit	2 CoComp	3 UDDT	4 AnDig
compost	-	654±*213*	662±*217*	526±*162*
Biogas	-	-	-	*690±157*

\- *not applicable*

5.2.3 Nutrient recovery

Currently there is no recovery of nutrients as fertiliser in Arba Minch. The introduction of co-composting and urine separation as proposed in systems 2 to 4 however, allows the recovery of nitrogen, phosphorus and other nutrients[118] for agricultural purposes. Table 5.4 shows the varying potential for nutrient recovery in the four systems. Relating the recovered flows to the total nutrient output (i.e. excreta, greywater, organic waste and manure) that is theoretically available for fertilisation, reveals that in the different systems only 7 to 17% of the total nitrogen and 13 to 25% of the total phosphorus is recovered (Figure 5.7). Therefore, the systems could still be improved to increase the

[118] As discussed in Section 3.2.1, potassium and sulphur are not included in the ceMFA model of Arba Minch. Yet, a qualitative assessment of these nutrients in the flows urine, faeces and organic waste shows that the recovery potential for K and S is analogue to N and P. This means that system 3 UDDT shows the highest potential for potassium and sulphur recovery, but also in system 2 CoComp and 4 AnDig these two nutrients are recovered for use in agriculture.

recovery rate. Although potassium and sulphur are not modelled for the Arba Minch case study, the possible recovery of these two nutrients is expected to be in the same degrees as nitrogen and phosphorus for the different systems.

Table 5.4: Specific recovered nutrient loads [kg p^{-1} y^{-1}]

	1 CuSit	2 CoComp	3 UDDT	4 AnDig
Nitrogen	-	0.99±*0.26*	2.31±*0.28*	1.89±*0.34*
Phosphorus	-	0.26±*0.07*	0.49±.*0.07*	0.45±*0.07*

\- *not applicable*

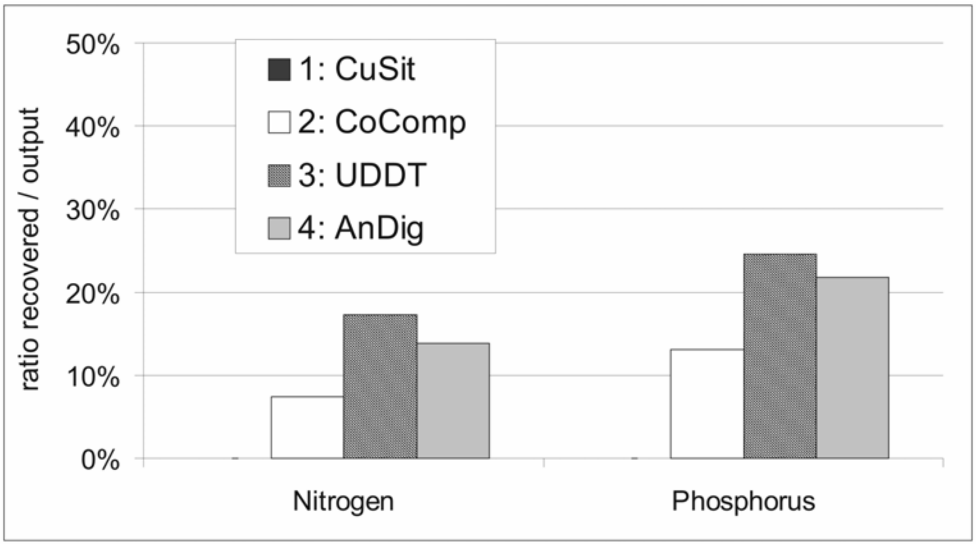

Figure 5.7: Ratio of recovered nutrient loads to total nutrient outputs (urine, faeces, greywater, organic waste and manure) in the different systems

The specific recovered loads can be converted into agricultural area that could be fertilised by the total nutrient flow using nutrient application rates shown in Section 5.1.1. This translates into areas between 2,000 ha (2 CoComp) and 4,700 ha (3 UDDT) for nitrogen and 1,000 ha (2 CoComp) and 2,000 ha (3 UDDT) for phosphorus (see Figure 5.8). These areas are 3% to 16% of the total agricultural area that is needed to feed the population of Arba Minch.

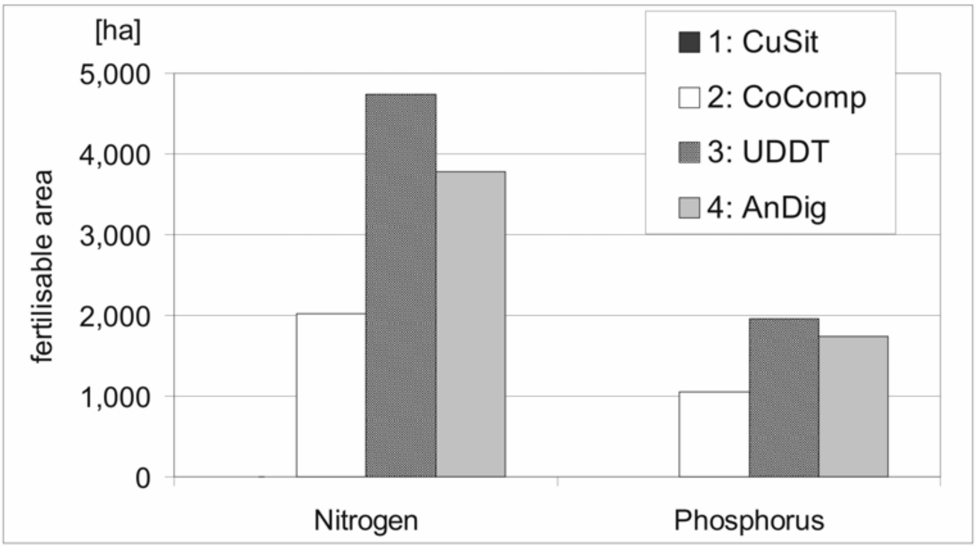

Figure 5.8: Nutrient recovery potential expressed as area that could be fertilised [ha]

5.2.4 Mass and water flows

Groundwater extraction for drinking water purposes is roughly about $1.1*10^6$ m^3 y^{-1} for systems 1 to 3. This volume increases by 17% to about $1.3*10^6$ m^3 y^{-1} for System 4 AnDig, where pour-flush toilets are introduced to flush excreta to the anaerobic treatment. One option to reduce the additional groundwater extraction is the introduction of rainwater harvesting. Assuming a roof collection area of $4m^2$ on every house and a runoff coefficient of 0.8, a volume of about 45,000 m^3 y^{-1}, i.e. 24% of the increased demand, could be covered by the collected rainwater. Storage[119] however, would be relatively costly at about 49 ETB p^{-1} y^{-1}. This is a relatively high cost that would need to be compared to the cost of water from other sources.

Urine diverting toilets as in System 3 UDDT usually cut down on water consumption. In System 3 however, the number of existing flush toilets is not changed in comparison to the current situation, but it is assumed that only those people who practice open defecation or use pit latrines will switch to the use of UDDT. The introduction of greywater recycling in 50% of all households, as included in systems 3 and 4, allows for the recovery of about 32% of the total extracted water. This water, however, would only be available on-site, since conveyance in pipes is not appropriate; particularly subsistence farming would benefit from this. Another recovered water flow is the pond effluent of the university's waste stabilisation ponds, which provides maximum about

[119] For the calculation a storage period of 6 months is assumed. The cost of the plastic rainwater storage tanks is assumed to be 1,700 ETB m^{-3} at a useful lifespan of 15 years and an operation and maintenance cost of 2% of the investment (see also Section 5.1.8).

$0.1*10^6$ m^3 y^{-1} to nearby agricultural fields. Compared to the total amount of irrigation water that is required (i.e. about $100*10^6$ m^{-3} y^{-1}) this flow is insignificant.

The source separation and recycling of waste and wastewater flows requires transport of the flows to or from treatment processes. Table 5.5 lists the total flows that need to be transported by a combination of donkey carts (collection in narrow streets) and lorries (transport from collection stations to treatment points). Particularly System 4 AnDig shows high transport requirements for faecal sludge and compost. The impacts of this mass flow analysis in terms of energy requirements and costs are discussed in the next sections.

Table 5.5: Mass flows requiring lorry-based transport [1,000 t y^{-1}]

	1 CuSit	2 CoComp	3 UDDT	4 AnDig
organic waste to co-composting	-	8.1±*2.2*	8.1±*2.2*	*5.6±2.0*
faeces to co-composting	-	-	8.2±*1.2*	-
faecal sludge to co-composting	-	7.2±*1.0*	0.4±*0.1*	229.1±*35.5*
manure to co-composting	-	7.0±*1.0*	8.4±*1.2*	-
compost to agriculture	-	9.8±*1.6*	11.5±*1.8*	55.9±*17.3*
urine to agriculture	-	-	21.1±*2.9*	-

\- *not applicable*

An inspection of the mass flow balance of the co-composting process reveals that the recommended mixing ratio of 2:1:1 (organic waste: cow manure: faecal matter) (see Section 5.1.8) cannot be adhered to. Although it is assumed that 100% of the collected organic waste from the markets and the households are added to the co-composting process, the ratio of organic waste to cow manure to faecal matter is roughly 1:1:1 in systems 2 and 3[(120)]. Additional organic matter such as organic waste from parks, landscaping and greening, which are flows that are not included in the ceMFA model, is recommended to be added to the co-composting process. This is also confirmed by a look at the quality of the compost in terms of carbon content and the C/N ratio. For all the systems both these values are rather too low; this emphasises the need for additional carbon input to the composting process. Regarding System 4 AnDig, dewatering of the faecal slurry added to the co-composting process should be further improved by for example planted drying beds. A detailed analysis of the composting process, however, is beyond the scope of this study.

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[120] In system 4 AnDig, where organic waste is added mainly to the digestion process, input to the composting process is almost exclusively slurry and the process will need to be adapted accordingly.

5.2.5 Energy analysis

The energy analysis of the four systems goes beyond the water, waste and wastewater systems, and includes the use of fuelwood and charcoal. This is done to reflect the benefit of biogas generation, which is currently only used for household purposes such as cooking, and can therefore also replace the use of fuelwood and charcoal for such purposes. Further processes that are included are water supply, transport and mineral fertilisers. An overview of total energy demands and the contribution of the different processes is given in Figure 5.9 and Table 5.6. The water supply of Arba Minch University is insignificant when compared to the water supply required for the rest of the town. The energy requirement for the provision of mineral fertiliser is between 187 kWh p^{-1} y^{-1} (3 UDDT) and 219 kWh p^{-1} y^{-1} (1 CuSit), depending on the nutrient recovery rate. Transport requirements play a role only in System 4 AnDig, where large amounts of faecal sludge and compost need to be transported. The increased energy requirements however, are offset by the biogas that is produced in the anaerobic digestion processes[121]. The biogas produced from wastewater, organic waste and manure can replace about 45% of the current energy obtained from fuelwood and charcoal.

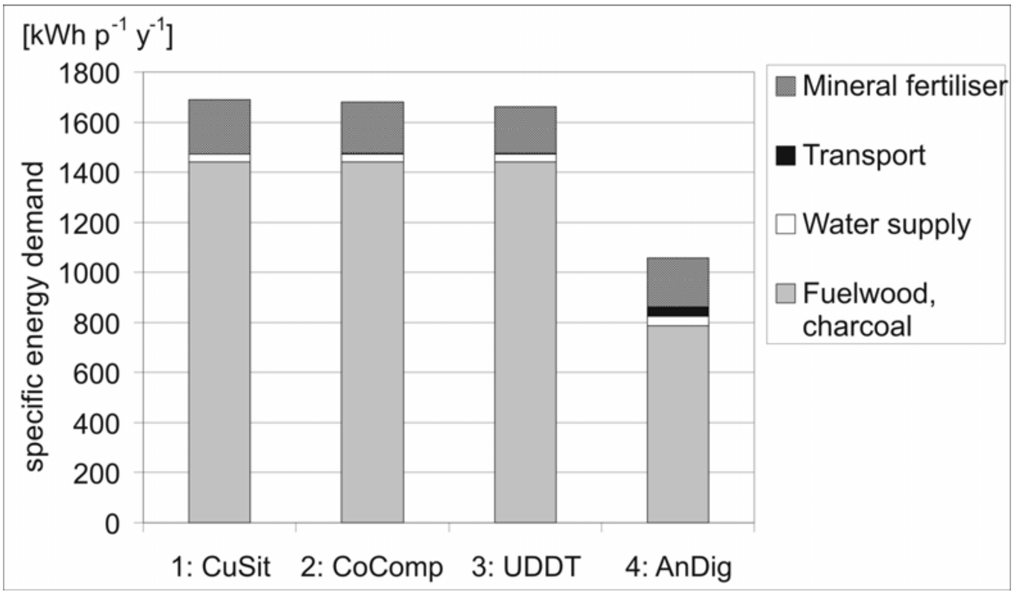

Figure 5.9. Specific primary energy demand [kWh p^{-1} y^{-1}]

The overall reduction in energy requirements for System 4 AnDig as compared to the current situation, amounts to about 38%. The other systems show a relatively similar energy demand. If, however, only the processes water supply, transport and mineral

[121] It must be noted, though, that this replacement is occurring at household level, whereas transport energy demand depends on other sources. The use of biogas for lorry fuelling as discussed for Hamburg (see Section4.2.5) does currently not seem to be feasible in Ethiopia.

fertiliser are taken into consideration, energy requirements for 4 AnDig increase by about 7% when compared to System 1 CuSit due to the high transport requirements. In this case, i.e. neglecting the use of fuelwod and charcoal, System 3 UDDT would be the most energy efficient system, yielding an energy reduction of about 12% as compared to the current situation; this is due to the replacement of mineral fertiliser.

Table 5.6: Primary energy demand per capita for the different processes [kWh p^{-1} y^{-1}]

	1: CuSit	2: CoComp	3: UDDT	4: AnDig
wood fuels	1440±*217*	*1440±217*	*1440±217*	*788±377*
water supply	32±*9*	32±*9*	32±*9*	*38±10*
transport	-	2±*1*	2±*1*	38±*13*
mineral fertiliser	219±64	205±62	187±60	193±61
total	*1691±290*	*1680±289*	*1661±287*	*1056±461*

- *not applicable*

5.2.6 Economic evaluation

The economic assessment is initially done without considering any benefits from recovery of nutrients or organic matter; this is done because urine and compost from human waste are not yet fully accepted as common merchandise. As a second step, the effect of financial benefits of urine, compost or biogas is discussed.

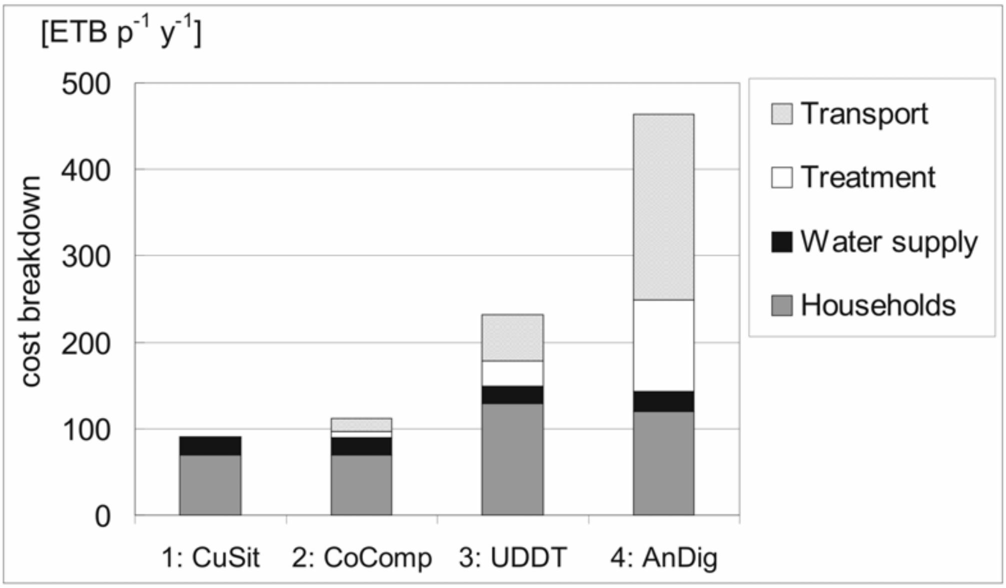

Figure 5.10: Cost breakdown [ETB p^{-1} y^{-1}]

The calculated costs related to the processes households, water supply, treatment/storage and transport are based on mass, nutrient and energy flows and the cost parameters listed in Section 5.1. Figure 5.10 shows the resulting annualised costs per person and the respective cost breakdown. The detailed cost items are listed in Table 5.7. Total costs are in the range of 90-110 ETB p^{-1} y^{-1} for systems 1 and 2, 230 ETB p^{-1} y^{-1} for System 3 and 460 ETB p^{-1} y^{-1} for System 4.

Costs of the Process Households include costs of on-site sanitation facilities and on-site greywater treatment. Total household costs are in the range 70 to 130 ETB p^{-1} y^{-1}. Comparing these numbers with the maximum costs that are perceived as affordable (i.e. 46 ETB p^{-1} y^{-1}, see Section 2.6), shows that even current sanitation options are rather expensive in the country's context. Implementation of UDDT or pour-flush toilets connected to on-site anaerobic digestion increases costs for the households by 70 to 80%. Greywater treatment amounts to about 20 ETB p^{-1} y^{-1} at an implementation rate of about 50%. The cost of water supply to public standpipes, house connections and AMU, is about 20 ETB p^{-1} y^{-1}. This amount increases by about 15% if pour-flush toilets are introduced on a large scale as in System 4 AnDig.

Treatment costs include storage of source separated urine and the co-composting process in systems 2 to 4. The analysis shows that costs of co-composting increase significantly in System 4 AnDig where large amounts of waste flows are treated. By comparison co-composting costs in Systems 2 and 3 are rather marginal. Urine storage costs about 21 ETB p^{-1} y^{-1}.

Table 5.7: Specific costs of the different processes [ETB p^{-1} y^{-1}]

	1: CuSit	2: CoComp	3: UDDT	4: AnDig
household				
pit latrine	63.7±*12.0*	63.7±*11.9*	-	-
septic tank	6.2±*0.8*	6.2±*0.8*	6.2[(122)]±*0.8*	-
UDDT	-	-	102.3±*14.9*	-
anaerobic digestion	-	-	-	100.0±*14.1*
greywater treatment[123]	-	-	20.3±*10.5*	20.3±*10.5*
water supply	20.2±*4.4*	20.2±*4.4*	20.2±*4.4*	23.3±*4.7*
treatment				
urine storage	-	-	21.1±*2.8*	-
co-composting	-	6.8±*0.7*	8.1±*0.8*	104.9±*13.9*
pond AMU	0.2±*0.0*	0.2±*0.0*	0.2±*0.0*	0.2±*0.0*
transport				
lorry	-	12.6±*3.8*	45.9±*12.7*	163.2±*58.7*
donkey cart	-	2.0±*0.6*	7.1±*1.9*	51.9±*15.6*
total	90.2±*13.1*	*111.8±13.7*	*231.4±24.9*	*463.8±73.0*

\- *not applicable*

The costs analysis shows that one of the critical economic factors is the Process Transport. While the cost of transporting pit contents and organic waste to co-composting (System 2) is still in a relatively small range, this cost increases when urine needs to be transported (System 3). System 4 shows highly increased transport costs making up about 50% of the total system costs.

In order to analyse the effect of a varying interest rate a parameter variation is carried out. The results show that generally the interest rate has little impact on overall costs since the ratio of large investments for infrastructure is rather small when compared to total costs. In general, an interest rate of 10% increases overall costs by only about 1% as compared to an interest rate of 0%. Since the different systems show a similar relationship, the results are not depicted here.

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122 This value represents the ratio of the population, i.e. 6%, that still uses septic tanks.

123 The relatively large uncertainty in this value derives particularly from the high uncertainty of the parameter *lifespan* of the greywater treatment facility, for which not enough experience has been gained yet.

Systems 2 to 4 allow the recovery of nutrients and organic matter for agricultural purposes. Therefore, benefits in terms of reduced mineral fertiliser application and the sale of compost, can be expected. In addition, the use of biogas can replace fuelwood and charcoal for household energy purposes. The allocation of financial benefits[124] using the cost parameters explained in Section 5.1.1 results in the potential revenues listed in Table 5.8. Especially the current high price of compost[125] leads to high benefits in all systems and in particular in System 4 AnDig, where large flows of digested slurry, manure and organic waste are co-composted. The benefit of urine application as a fertiliser is about 20 ETB p^{-1} y^{-1}, and therefore compensates for the cost of urine storage, but does not give any additional benefit. The use of biogas yields a benefit of about 60 ETB p^{-1} y^{-1}, which fully covers the additional costs of the biogas plants.

Table 5.8: Benefits from recovery of nutrients and organic matter [ETB p^{-1} y^{-1}]

	2: CoComp	3: UDDT	4: AnDig
benefit urine	-	19.3±*3.2*	-
benefit biogas	-	-	59.8±*12.0*
benefit compost	184.4±*34.4*	216.4±*40.6*	1048.2±*342.6*
benefit total	184.4±*34.4*	235.7±*40.7*	1068.1±*349.1*

\- *not applicable*

Relating the expected total benefits to the total costs of the systems results in the benefit-cost ratios depicted in Figure 5.11. A benefit-cost ratio greater than one shows that the expected benefits outweigh the costs of a system. System 3 UDDT shows about equal costs and benefits, whereas the benefits from compost sale more than fully recover the costs of systems 2 CoComp and 4 AnDig. It can however be questioned whether the current high price of compost will still be valid if supply increases, or if a decline in price occurs. Therefore, a variation of the compost price is done in Section 5.3.2.

[124] Benefits of improved sanitation systems for health and environment are not assigned monetary values. For an overall assessment however, these benefits should be taken into consideration.

[125] Please note that the financial benefits of compost are based on weight only. As mentioned in Section 5.2.4 the quality of the compost is rather low because of low organic carbon content. Therefore, it is doubtful whether the current high price can be achieved in the long term, without adaptation of the composting process by, e.g. addition of supplemental flows rich in organic carbon.

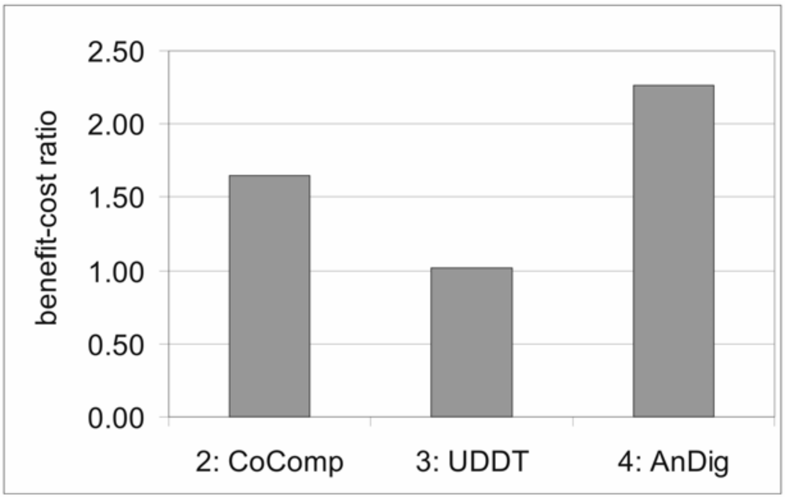

Figure 5.11: Benefit-cost ratios of Systems 2 to 4

5.2.7 Discussion of the results

The main results of the above sections are summarised in Figure 5.12. The introduction of the processes for nutrient and organic matter recovery can decrease the use of mineral nitrogen fertiliser by between 7% (2 CoComp) and 16% (3 UDDT). Mineral phosphorus fertiliser use can be decreased by about 3% (2 CoComp) to 6% (3 UDDT, 4 AnDig). The differences in these two reduction rates derive from different nutrient ratios in the considered waste flows as well as from different current application rates of mineral fertiliser[126]. Therefore, the addition of phosphorus fertiliser to make up for the difference, needs to be considered.

Groundwater extraction is only affected in System 4 AnDig, where pour-flush toilets are installed and total water demand is increased by 17%. Recycled greywater is not considered to reduce the overall water demand, since its use will be primarily for additional irrigation purposes, and not replace the tap water currently used for household purposes. Marginal energy reductions can be observed in Systems 2 CoComp and 3 UDDT due to the replacement of mineral fertiliser. In System 4 AnDig however, where biogas is used to replace wood fuels, the overall energy balance shows a reduction of more than 37%.

The economic assessment depends greatly on whether monetary benefits are considered or not. If only costs are looked at, Systems 3 UDDT and 4 AnDig appear to score poorly with cost increases of 156% and 414% respectively (see Figure 5.12). System

[126] A comparison with the Hamburg case study, where relatively more phosphorus fertiliser is replaced by human waste products, shows that current phosphorus application in Ethiopia is comparatively higher whereas the recommended nitrogen application rate is less than in Germany (see also Sections 4.1.1 and 5.1.1).

2 CoComp shows a cost increase of 24% compared to System 1 CuSit. Yet, if potential benefits from the sale of compost and urine and the reduced expenses of wood fuel are taken into consideration, total system costs can be offset in all three alternative systems. This is shown by the benefit-cost ratios in Figure 5.11. A more detailed financial analysis would therefore be needed, that looks into the potential of making a financial profit from waste products (such as compost) to support the assumptions that are used in this study.

All in all it can be expected that all three systems considered are beneficial in terms of resource efficiency, reducing energy and mineral fertiliser needs. In addition, the systems improve hygienic conditions, and therefore result in important benefits with regard to health and the environment. Eventually, the investment in sanitation infrastructure will also lead to economic benefits in terms of reduced illness and reduced loss of productive time.

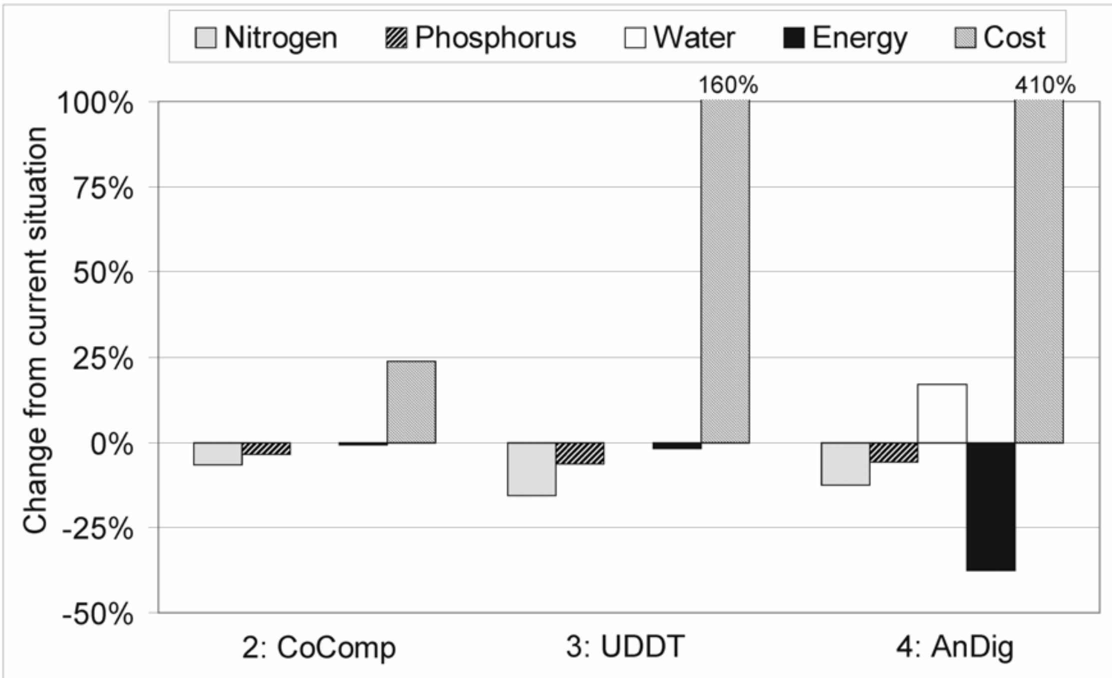

Figure 5.12: Potential change of mineral fertiliser use, groundwater extraction, energy demand and annualised costs compared to the current situation [%]

5.3 Sensitivities, parameter variations and system modification

The results discussed in the previous sections depend on the selected systems parameters and their variability. A sensitivity analysis and parameter variations, as they are presented in the next sections, can help in understanding the system behaviour. In addition, the model is used to analyse a possible combination of the three systems by the variation of those system parameters that define the specific sanitation systems.

5.3.1 Key parameters

A sensitivity analysis is carried out to identify the most sensitive parameters with regard to the criteria nutrient recovery, energy demand and costs. Those parameters that show a high relative sensitivity are summarised in Annex G. Regarding nitrogen and phosphorus recovery, the implementation rate of specific processes, such as UDDT, treatment of sludge, and slurry in the co-composting process, is a very important and sensitive factor. In addition, nutrient loads in faeces, manure and organic waste are parameters that should be defined with great care, since their values have a relatively high impact on the modelling results with regard to nutrient recovery. Also transfer coefficients of nitrogen and phosphorus in the composting process are parameters that have a relatively high sensitivity.

The most sensitive parameters for overall specific energy requirements are those parameters related to the energy demand from wood fuel and charcoal. Also those parameters that are used to define the energy requirements from nitrogen fertiliser application (e.g. agricultural yield, food intake per person, N application rate, etc.) show a high sensitivity. For System 4 AnDig, parameters that are related to biogas generation (e.g. biogas production per organic matter (as VS), energy yield from biogas), have a high inverse sensitivity. This means that if these parameters increase the total energy demand will decrease.

The sensitivity of transport cost parameters (i.e. cost per km, maximum distances and truck capacities) is important for the total costs of Systems 2 to 4. In addition, operation and maintenance costs, as well as the investment costs of pit latrines and UDDT, are sensitive parameters for the overall costs in systems 2 and 3. Total costs of System 4 are sensitive particularly towards parameters that influence volumes to be transported, for example, the amount of water used for flushing or the total volume of slurry added to anaerobic digestion processes. Household size shows an inverse sensitivity in all systems, i.e. specific costs decrease if household size increases[127].

For more details with regard to the sensitivity analysis please refer to Annex G.

5.3.2 Variation of selected parameters

The results of the ceMFA discussed in Section 5.2 are dependent on a large number of system parameters that are used to describe the material flow system. A change in these

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[127] For example, if the number of people per household increases from 5 to 6 (i.e. a 20% increase), the total specific costs in systems 2 and 3 decrease by about 11 to 12%.

parameters either due to actual developments or due to false estimations can result in significant differences in the results. A first step to assess these differences is the sensitivity analysis as carried out in Section 5.3.1. Another important way of analysing the results, is the variation of significant parameters and the assessment of potential changes.

To exemplify the possible variations of total costs, the parameter transport costs which is initially set to 15 ETB km^{-1} (see Section 5.1.7), is varied between 0 and 30 ETB km^{-1}. Figure 5.13 shows the resulting variation in costs for systems 2 to 4. As previously discussed System 4 AnDig in particular is greatly influenced by transport costs. A doubling of transport costs from 15 to 30 ETB km^{-1} increases total costs of System 4 AnDig by 35% and that of System 3 UDDT by 20%.

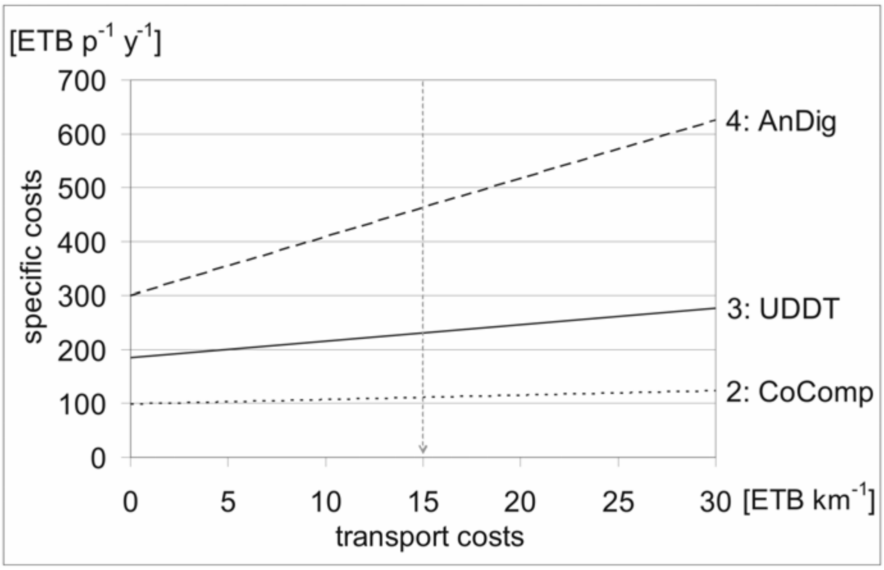

Figure 5.13: Impact of transport costs on total costs [ETB p^{-1} y^{-1}] (initial parameter marked in the figure)

As discussed above, systems 2 to 4 can achieve economic benefits by recovering nutrients and organic matter. Yet, the actual financial value of the benefits depends on market developments. For example, a saturation of demand for compost by a boost in production, can result in a drop in prices. A variation of the compost price shows the great impact of the compost revenue on expected overall benefits, particularly for System 4 AnDig (see Figure 5.12). In order to look at the critical compost price (i.e. the break-even price where costs equal benefits), an analysis of the benefit-cost ratio is done dependant on the compost price. This analysis reveals that the benefits of systems 2 and 4 are greater than the respective costs at a compost price greater than approximately 0.7 ETB kg^{-1}.

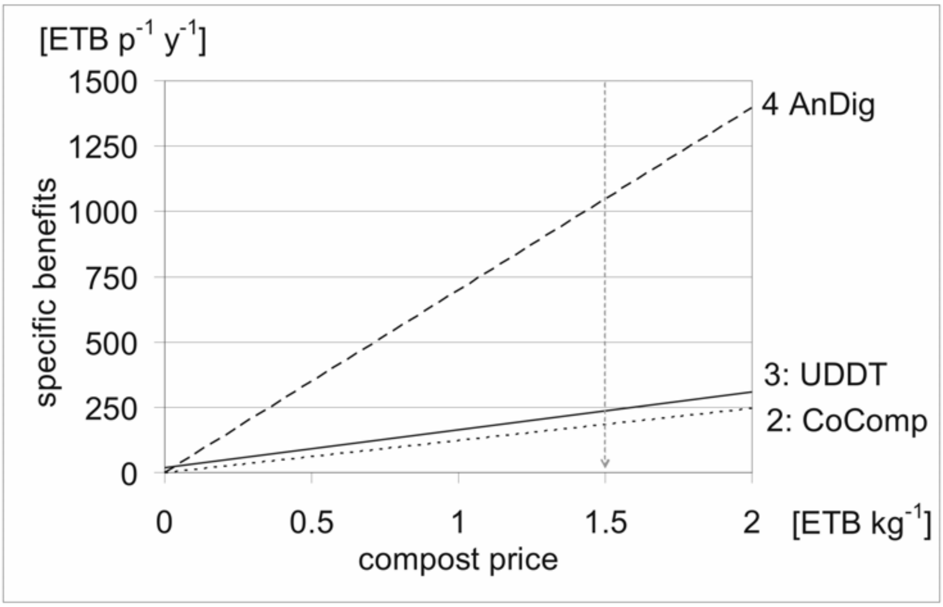

Figure 5.14: Impact of compost price on specific benefits [ETB p^{-1} y^{-1}] (initial parameter marked in the figure)

The addition of other organic matter to the anaerobic digestion process is of particular importance for System 4 AnDig. In the initial system analysis it is assumed that 20% of the manure and 60% of organic household waste is added to the biogas plants. To assess the impact of cow manure treatment in biogas plants on overall costs and energy balances, a parameter variation is carried out. Figure 5.15 highlights that energy production can be increased significantly, even exceeding total energy consumption, if more than 70% of the cow manure is added to anaerobic treatment[128]. At the same time, total costs increase due to increased transport and treatment requirements. Financial benefits from biogas use however, are not included in this value. Checking the benefit-cost ratio reveals that the additional benefits of biogas generation outweigh the increased costs. Furthermore increasing benefit-cost ratios are observed at increasing rates of cow manure digestion. For example, the addition of 80% of the cow manure to the biogas digesters compared to 20%, increases the benefit-cost ratio from 2.3 up to 2.5.

The cattle farming in Arba Minch however, is currently done as extensive farming and comprises mainly free ranging animals. This means that cows are sheltered in stables only at night-time. Therefore, in order to make full use of the additional biogas potential, there needs to be an increased collection of manure, e.g. in permanent stables.

[128] Please note that in this analysis the energy value of cow manure used as household fuel (i.e. dried dung cakes) is not included in the energy analysis. This is common practice in Ethiopia and also in Arba Minch, but it is considered to be very energy-inefficient (Seyoum, 1988).

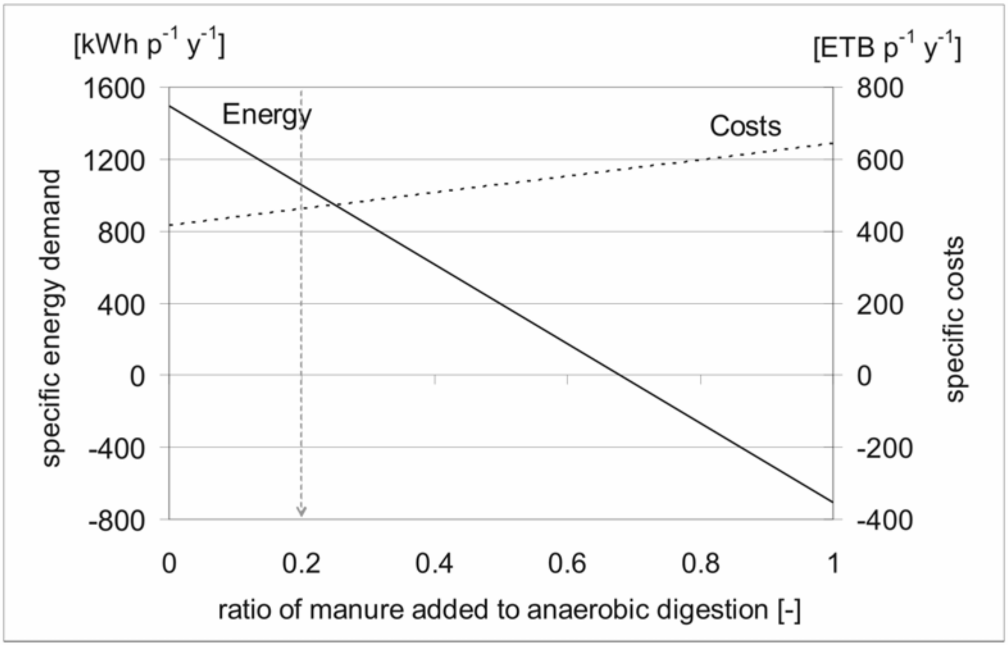

Figure 5.15: Impact of manure treated in anaerobic digesters on energy demand [kWh p^{-1} y^{-1}] and costs [ETB p^{-1} y^{-1}] of System 4 AnDig
(initial parameter marked in the figure)

5.3.3 Combination of the systems

The analysed systems represent only extracts of possible system developments and each focus on one specific technology. In reality however, future developments in Arba Minch will most probably be constituted by the implementation of a variety of different technologies, depending on the household priorities. This is also reflected by the Sanitation Business Plan, first developed by Arba Minch Municipality in 2009 (Esatu, 2009). According to this plan, the available budget is segmented into innovative sanitation technologies such as UDDT, and into improved conventional technologies such as septic tanks or co-composting of pit sludge. Therefore, the modelling is carried out for a combined system made up of a mix of sanitation technologies (see Table 5.9). It is assumed that about half of the population uses UDDT, and that 30% of the population uses either pit latrines and so-called fossa alterna, with a subsequent co-composting of the pit contents. The remainder of the population either continue using septic tanks (6%) or install small-scale anaerobic digesters (14%) for the treatment of blackwater, organic waste and cow manure[129].

[129] The number of households involved in dairy farming in Arba Minch is relatively small (Plückers, 2009). It is assumed that anaerobic digesters are only installed in those households that can use the digester for the treatment of cow manure as well.

Table 5.9: System parameters of the combined system (implementation rate in %)

	Combined system
type of toilet facility	
open defecation	-
pour-flush with anaerobic wastewater treatment	20
of which biogas plants are	70
UDDT	50
pit latrines / VIP / Fossa Alterna	30
organic waste from households	
given to livestock	30
collected	60
collected and added to co-composting	50
collected and added to biogas plants	50
organic waste from market	
added to co-composting	100
manure	
used as fuel	10
added to co-composting	-
added to biogas plants	20
greywater recycling	50

- *not applicable*

The model shows that in terms of nutrient recovery, the combined system is superior to all the other investigated systems. As a result of the nutrient recycling from urine and compost, more than 18% of the mineral nitrogen fertiliser and about 8% of the mineral phosphorus fertiliser could be replaced. An energy analysis shows a potential for saving about 27% of the energy demand as compared to the current situation. Water requirements (plus 3%) and cost requirements (293 ETB p^{-1} y^{-1}, i.e. plus 225%) are increased in the combined system when compared to the current situation. Expected benefits amount to about 40 ETB p^{-1} y^{-1} for biogas, 61 ETB p^{-1} y^{-1} for urine and 442 ETB p^{-1} y^{-1} for compost. Therefore, the benefits far outweigh the costs and make this system profitable.

This analysis shows that a system consisting of a mix of technologies, as the Arba Minch Sanitation Business Plan suggests, can provide a promising alternative in terms of resource efficiency and financial benefits. The ceMFA model can be used to assess further possible system developments (i.e. different ratios of applied technologies) and can be adapted according to actual developments. Therefore, the ceMFA model could be developed in line with the implementation of the Sanitation Business Plan and could be used to assess any projected interventions.

5.4 Transformation processes

Considering the current state of the sanitation infrastructure in Arba Minch, it is obvious that improvements and a system change are recommended. Apart from system enhancements by improved resource efficiency, external drivers may also play a role in system transformation. The following sections highlight the decision environment specifically for the case of Arba Minch, but also for Ethiopia in general. The systems under consideration are first scored against possible drivers for system change. Then, preconditions and starting points are discussed, as well as possible challenges identified.

5.4.1 Drivers for change

A system transformation might be initiated and supported by a variety of drivers. In the following, an overview is given of drivers considered to be important in the context of Arba Minch and Ethiopia.

- Dignity, comfort: The currently prevalent toilets, which are often unsheltered pits with wooden logs, can be linked with a lack of privacy and inconvenience due to their temporary structure. As economic development continues, it can be expected that households will be willing to invest in more "modern" toilets (e.g. in-house toilets) that provide more privacy, more comfort and a higher social status (Hernandez et al., 2009).

- Health: There is an established relationship between water and sanitation on the one hand, and specific diseases on the other[130]. These diseases can only be reduced if in addition to appropriate hygiene measures, sanitation infrastructure is improved. Although this might not be a primary driving force for households, public authorities often consider this as most important driver for improving sanitation.

- Population increase: Towns as Arba Minch are going to see a significant population increase in coming decades due to demographic change and migration from rural to urban areas. Therefore, the water and sanitation infrastructure needs to be adapted to increasing population density, and needs to achieve more effective treatment.

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[130] More than 60% of national disease burden in Ethiopia is attributed to waterborne and sanitation-related diseases (Tesfaye, 2008).

- Water scarcity: The availability of safe water resources is at risk considering an increase in water demand due to changes in population and specific water consumption. Therefore, sanitation systems that require as little water as possible are advantageous compared to water-hungry devices and systems.

- Availability of fuels: Household fuels such as wood fuel or charcoal are becoming scarcer and their exploitation is connected to deforestation and soil degradation. Electrical energy supply in Ethiopia is generally affected by disruptions and insufficiencies. Sanitation systems, therefore, need to be energy-efficient or even generate household fuels such as biogas.

- Soil degradation: Soil degradation and soil erosion is a widespread problem in Ethiopia and is also observed in Arba Minch. The addition of organic matter in the form of compost can contribute to healthy soils and prevent further soil degradation.

- Mineral fertilisers: Ethiopia's fertiliser supply is dependent on world market prices, which are often unaffordable to local farmers. Yet, in order to keep soils fertile and to yield good harvests, addition of nutrients is essential. Nutrients recycled from human waste products can represent a viable source of nutrients for agriculture.

In order to show how the analysed systems respond to the above-mentioned drivers and to analyse how they can be embedded into a wider economic, social and environmental framework, Table 5.10 illustrates an evaluation of the systems. The scoring indicates how well a system meets the respective drivers associated with the indicators mentioned in the table.

Table 5.10: Evaluation of the systems regarding possible drivers for transformation

(0 = system shows no advantage, + = system is beneficial, ++ = system is very beneficial)

	2 CoComp	3 UDDT	4 AnDig	*Indicator*
Dignity/comfort	0	+	++	*Type of toilet*
Health	+[a)]	+[a)]	+[a)]	*Emissions to environment*
Population increase	+	+	+	*Space requirements/ treatment effectiveness*
Water scarcity	+	+	0	*Water consumption see Section 5.2.4*
Availability of fuels	0	0	++	*Generation of energy see Figure 5.9*
Soil degradation	++	++	++	*C balance see Table 5.3*
Availability of mineral fertiliser	+	++	++	*N, P balances see Table 5.4*

a) All systems can achieve a reduction of pathogen release into the environment provided that proper handling and treatment are in place.

5.4.2 Preconditions

Political will is often an important precondition to support large scale improvements or transformations of the water and sanitation situation. In Ethiopia, the government has expressed this will through the Universal Access Plan, which aims for 100% sanitation coverage by 2012 (Ayenew, 2009). Further capacity development particularly on the local and regional level will be required to achieve this ambitious target. A coordinated approach bringing together the different stakeholders will be a further requirement. On the national level, first achievements have been made by WASH initiatives, including the Ministry of Water Resources, the Ministry of Health and others. Yet, on a local level responsibilities are often unclear, hindering a sustainable strategic planning process. Apart from the respective municipal departments (e.g. department of works and urban development, and the department of water resources or the local town water services), private sector organisations could potentially contribute to the operation and maintenance of water and wastewater systems. Additionally end-users such as farmers associations are important members of the planning process. If reuse is to be achieved on a large scale, agricultural stakeholders on a national and regional scale will need to be involved and contribute towards a positive perception of excreta-based fertilisers; this could be achieved by regulations, recommendations, incentives and encouragement by the local and national authorities.

Another important precondition is to develop a positive perception towards sanitation in general and towards resource-oriented sanitation in particular, among local households. Information and lobbying campaigns are needed to make sanitation a household priority in the light of tight household budgets. For Arba Minch in particular, the identification of suitable agricultural areas[131] that can receive the urban nutrient and compost output, is an important issue. Access to these areas and the associated logistics need to be considered before establishing a larger reuse scheme.

5.4.3 Starting points

In consideration of public hygiene and health, the Sanitation Business Plan of Arba Minch targets first those households that currently do not have a toilet or own only poor pit latrines (Esatu, 2009). This group makes up about 30% of the households in total. But in this case the financial capacity of the households needs to be considered and subsidies might be needed for implementation. Another approach is to look for the so-called "early adopters", i.e. persons willing to try innovative sanitation technologies because of problems with current designs in rocky soils, or areas prone to flooding, or because of status. Additional areas in Arba Minch that could be targeted are expansion areas where new houses are built to accommodate the growing population. Last but not least, those neighbourhoods where innovative sanitation technologies have been implemented as scattered pilot units within the scope of previous projects (such as ROSA), have already been exposed to new technologies and might be more open to adopt these. In general, the approach of initiating a system transformation should rather be demand-driven than supply-driven, to ensure that appropriate participation and a feeling of ownership develops.

In the context of Ethiopia, where a change in sanitation systems is strongly linked to household behaviour and perceptions, social marketing might be a useful approach. One approach that is being more frequently used in Ethiopia is the so-called Community-Led Total Sanitation (CLTS) approach using the concepts of shame and disgust to ban open defecation and improve the hygienic situation of communities (Kidanu and Abraham, 2009). Traditional community-based organisations might also have a role to play in promoting hygiene and sanitation, and to find ways of transforming the current water and sanitation systems. Schubert (2008) analysed community-based organisations in Arba Minch and identified cooperation options to achieve more resource-oriented sanitation. For example, youth groups could be

[131] The agricultural areas within the town borders are not sufficient to accept the nutrients if a recovery system is implemented in full scale. However, there is a diverse range of agricultural activities in the rural areas surrounding the town of Arba Minch.

supported to get involved as stakeholders in different processes such as collection and transport of waste flows, compost preparation or in constructing and maintaining sanitation facilities.

Regarding agricultural reuse of nutrients and organic matter the Arba Minch state farm could be a suitable starting point. Changes in ownership and the lack of application machinery, have resulted in a unenthusiastic response from the state farm up to now despite good results in crop trials (see, for example, Ercolano, 2009). Therefore, the agricultural micro and small enterprises within the town borders, as well as farmers in the proximity of Arba Minch, who struggle with depleted soils, represent promising starting points for agricultural use of urine and compost. All in all, including the private sector at various stages in the waste recovery scheme, seems to be important for a sustained operation (Drewko and Otterpohl, 2009).

5.4.4 Challenges

A system transformation, including the reuse of waste flows needs to be prepared to counter several potential challenges and obstacles. Particularly public perceptions and acceptance need to be shaped by raising awareness, information and implementing trials. These efforts should be targeted at a local level, and also on higher levels, where often more preconceptions with regard to reuse exist.

Several stakeholders and organisations play important roles for the creation of a sustained value chain. Considering the diversity of groups, decision makers and interests in Ethiopia, this could represent a potential stumbling block for the continued running of a system. Therefore, strategic planning that includes backup plans and redundancies[132] is considered to be important.

The financial benefit of a system transformation depends on the recognition and approval of the beneficial effect, i.e. using organic matter from waste as source for energy and soil conditioner and using nutrients for improving soil fertility, and its monetary valuation. Therefore, reasonable market prices for human waste products are important to compensate the higher investments for recycling systems. Nevertheless, households and public bodies may be put off by high investments if there are no direct benefits; affordable loans or other financial sources therefore need to be available.

[132] For example, to prevent a collapse of the system in case a part of the value chain fails, collection of waste flows should not only be done by one service provider.

6 Concluding discussion

In the following sections the method and the results of this study are set into a broader context. Firstly, potentials and possible drawbacks of the selected approach and in particular of the ceMFA model, are discussed. Secondly, general results of the assessment of the two case studies are reviewed from an overall perspective, going beyond the discussion of particular results as presented in sections 4 and 5. The chapter closes with some remarks regarding resource efficiency in decision making.

6.1 General approach and applied method

Resource efficiency assessments in sanitation can contribute to strategic infrastructure planning and decision making. To date, standardised ways for defining and assessing the resource efficiency in sanitation are still lacking. Material Flow Analysis is a useful method to analyse different sanitation systems in a systematic and transparent way. The combination of mass, nutrient and energy flows with cost estimates in a ceMFA – as developed in this study – provides a valuable tool for integrated assessments. The systems perspective taken up in the ceMFA is considered to be essential for a holistic analysis, as opposed to a focus on single technologies or facilities. The following list highlights general conclusions regarding the modelling process:

- The combination of mass and nutrient flows (i.e. physical flows) as well as energy demand and costs in one model, reflects the interdependency of the considered assessment criteria. Thus the environmental as well as the economic performance can be evaluated simultaneously. Trade-off relationships can be identified and a holistic assessment can be carried out.

- Understanding of the sub-systems, which make up a complete sanitation system, is greatly improved by the model. Unfavourable system components (i.e.

processes or flows) that impair the overall efficiency, can be identified and can be improved accordingly. Also, required process adaptations can be specified[133].

- Modifiable system and process parameters that have the greatest impact on resource efficiency and economics can be identified[134]. When it comes to the actual implementations, there should be a special focus on these parameters.

- Transition options, as well as combined (hybrid) systems comprising several different technologies, can be modelled if system equations are set up accordingly. This allows for assessments which reflect more realistic sanitation scenarios and can help to identify the optimum system.

- Material flow schemes derived from the modelling can be used to provide a quick overview of the system behaviour, highlighting important flows and processes. Different systems can easily be compared at a glance.

- Data quality and appropriate assumptions are crucial. Uncertainty analyses help in understanding the influence of uncertainties upon the results.

- Sensitivity analyses and parameter variations allow checking the variability of the results. In addition, parameter variations are a useful tool to identify if there is any room for improvements or to simulate possible future developments.

- The results of the modelling should not be seen as the means to an end; they should rather be viewed as a sound basis for a decision making process, which includes the discussion of objectives and the development of systems suitable for achieving these objectives. Prioritising and weighting of the modelling results might, therefore, be required for multi-criteria decision making.

The ceMFA model has been specifically developed for the case studies of Hamburg and Arba Minch; as well as the alternative systems selected. The systems have been selected to represent a wide range of possible options, with a varying degree of source separation and a varying degree of centralisation. Additional systems can be assessed using the developed ceMFA model. For example, in the case of Hamburg, technologies such as the recycling of toilet wastewater, i.e. the "LooLoop" (Braun et al., 2008), or the

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[133] For example, the modelling shows that the co-composting process for the case study Arba Minch would require additional carbon-rich inputs.

[134] Examples for such variables include the volume of flush water or the collection rate of organic waste.

implementation of kitchen waste grinders[135] could be modelled and assessed. For Arba Minch, a system based on treatment of wastewater by bamboo[136] could be a possible resource efficient alternative.

With the necessary modifications, the ceMFA model can equally be applied to other case studies and their specific conditions. This would require a thorough systems analysis of such a case study, and subsequent modifications of the mathematical model equations where necessary. In addition, those parameters, which are dependent on site-specific conditions such as persons per household, need to be adapted accordingly. A good knowledge of the peculiarities of the urban systems under consideration, as well as a good understanding of the modelling procedure, is a prerequisite for the successful transfer of the model to other case studies.

The focus of this study is on certain criteria (e.g. nutrient recovery) and indicators (e.g. N, P) for resource efficiency in sanitation. Yet, the model can be extended to also include additional criteria that are important for decision making. For example, the fate of pathogens, micropollutants or greenhouse gas balances could be linked to physical flows in the model. However, detailed and specific knowledge about the behaviour of these parameters would be required to enable proper integration of these criteria into the ceMFA model[137].

It must be remembered that a mathematical model such as the ceMFA is only as reliable as the underlying data. Therefore, data uncertainty is of great concern, particularly in cases such as developing countries where reliable data is scarce. This emphasises the need to carefully evaluate, select and estimate appropriate data, and to analyse the effect of data variability on the results. The software SIMBOX used for the ceMFA, allows state-of-the-art analysis of data variability by first-order uncertainty analyses and Monte Carlo simulations. The identification of uncertainties and sensitivity analyses can help to improve the quality of the results by improving the input data.

[135] Kegebein (2006) analysed the functioning of kitchen waste grinders in Germany and concluded that this represents a technology that compares favourably to the composting of organic waste, but that from an energy perspective, is less efficient than anaerobic digestion of organic waste.

[136] For an introduction to wastewater treatment by bamboo, please refer to, e.g. Ndzana and Otterpohl (2009).

[137] For example, Leinemann (2008) used a material flow analysis to determine inputs of pharmaceuticals excreted via urine to agriculture. The behaviour of pharmaceuticals in soil and plants, however, could not be modelled satisfactorily.

The results depend not only on data quality, but also on selected system boundaries. Therefore, it is important to consider the results within the framework of the temporal and spatial system boundaries. Firstly, temporal variations, i.e. dynamic system behaviour, are not within the scope of this study. But for planning it could be useful to include system transformations into the model (i.e. to model the stepwise dynamic implementation of certain technologies towards a full system transformation) for an assessment of changes in material, energy and costs flows over time. Secondly, spatial system boundaries are defined by administrative boundaries. Any effects referring to external processes (e.g. nutrient emissions to surface waters) are indicated by the respective output flows. Economic assessment does however not include any externalities or intangibles, but focuses on direct and indirect costs, such as investment, and operation and maintenance, occurring within the system boundaries. The inclusion of external costs and benefits, such as river pollution downstream or employment generation, and intangibles such as improved quality of life, could possibly shed a different light on the economic assessment. These factors are important criteria for decision making and should be further investigated by a detailed economic analysis, putting some sort of value on these costs and benefits that are difficult to quantify. Another important issue with regard to the economic analysis is the timing of costs and benefits. To consider this, a dynamic modelling approach could be more useful.

System boundaries also need to be considered for the interpretation of the energy analysis. Since preceding production processes are not included, the energy values in this study are only referring to actual operational energy consumption or generation, but do not include the so-called grey energy or cumulative energy demand of physical installations. The difference may be important in the case of pre-existing infrastructure. In contrast to sunk cost, which should be neglected for any rational economic decision making, the “sunk energy”, i.e. the energy that already went into the construction of available facilities, might influence future decisions. Also for decisions regarding new developments, i.e. greenfield developments, energy demand for production processes differ particularly for systems relying on multiple piping networks and additional installations[138].

6.2 Towards more resource efficiency in sanitation

Modelling the resource efficiency of different sanitation systems for two urban setups in different contexts, allows comprehensive insights and conclusions. The next two

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[138] Remy and Ruhland (2006), who carried out an LCA of source separating sanitation systems, concluded that the energy demand for construction, which makes up about 10% of the overall energy demand, may increase up to 60-80% as compared to a conventional system.

sections highlight some overall results of the ceMFA and discuss the implications for decision support.

6.2.1 Results of the modelling

The analysis shows that there is a variety of different systems available that allow the recovery of nutrients from wastewater flows and that suit the specific conditions. The selected systems represent only an extract with many more possible variations imaginable, but are considered to reflect some basic options. For both case studies the ceMFA shows that benefits in terms of resource efficiency can be gained, but that none of the systems is advantageous in all criteria considered.

The decision as to which system should be selected, depends on the specific objectives and priorities. For example, if overall energy efficiency should be increased, than Systems 3 NuRU (Hamburg) and 4 AnDig (Arba Minch) are the most favourable. If the primary aim is phosphorus recovery, than Systems 5 BlaD (Hamburg) and 3 UDDT (Arba Minch) achieve the highest recovery rates. If, however, the current system should be adjusted to include nutrient recovery at the lowest effort in terms of alteration of facilities and additional costs, than Systems 2 NuRS (Hamburg) and 2 CoComp (Arba Minch) would be the preferred systems.

In addition, some of the systems achieve benefits in several criteria, such as nutrient recovery plus saving of water, which can be valued accordingly. Source separation of wastewater can also show additional positive effects such as increasing the capacity of the existing wastewater infrastructure or reducing the emissions of micropollutants, which might be a plus factor in decision making.

For the analysis of the modelling results it is important to consider the specific underlying assumptions. The parameter variations showed that a special focus should be on certain modifiable parameters that have a large impact on the overall results (sections 4.3.2 and 5.3.2). For example, energy efficiency of those systems that include anaerobic treatment processes can be significantly improved if the flush volume is further reduced or if more organic waste is added. Therefore, further optimisation of sub-processes, as exemplarily shown in Section 4.3.3, is important to optimise the systems as a whole.

The modelling of a hybrid system for Arba Minch, i.e. a system comprising several combined system elements (see Section 5.3.3), shows that the interplay of different technologies can contribute to an overall performance improvement. This has not been studied for the case of Hamburg, since the model equations are set up in a different way to those for Arba Minch. It would also be interesting to check how partial implementations of different technologies interact in the case of more advanced wastewater infrastructure.

Not only can the ceMFA show potential benefits, but it can also identify possible drawbacks or trade-offs. In most of the source-separating systems transport by lorry (or donkey cart in Arba Minch) is required, which represents an additional cost and energy factor. The analysis shows that in general the energy balance of transport of source separated urine is acceptable as long as the distance between collections (i.e. urban areas) and use (i.e. agricultural areas) are within a reasonable range[139]. For larger volumes such as blackwater however, preceding volume reduction of the liquid (e.g. by dewatering) should be considered. Alternative fuels such as natural fuels or biogas should be assessed for their suitability. In addition to the energy perspective, other issues such as traffic congestion could be a point of failure when considering the lorry based transport of waste flows in urban areas.

Also, nutrient recovery processes can impact on the overall resource efficiency of the systems. Many processes such as the stripping of urine or the thermal pasteurisation of blackwater, show a considerable energy demand. Identifying and making use of synergies, such as using waste heat from other processes or combining the treatment of organic waste and human waste flows, is crucial for optimising these processes and improving the overall energy efficiency of the systems.

One further drawback is the anticipated cost increases linked to a system change (see Sections 4.2.6 and 5.2.6). Yet, the results of the ceMFA should not be overrated and should not be used for ruling out certain systems, since the results depending on the quality of the input into the model, could represent mere ballpark figures. The model should rather be used to detail the analysis using more precise cost data and using more accurate quantifications, once priorities for system selection are set. The importance of the right assessment framework is highlighted by the difference between the costs for system implementation in an existing infrastructure and in a greenfield development. If infrastructure facilities that are already depreciated (e.g. sewer systems) are not available, then the implementation of alternative systems may represent more cost-efficient alternatives than the conventional system[140]. For the case of Germany the difficulty will be to properly estimate the extensive rehabilitation requirements for the ageing wastewater infrastructure, which actually represents the case of an intermixture

[139] For the case study Hamburg, the break-even distance, which is the distance that source separated urine can be transported without undoing the energy gains at the WWTP, is calculated to be about 90 km.

[140] Regarding Hamburg, this is the case for systems 3 NuRU and 6 CompU.

of existing and new infrastructure. In this case a step-wise implementation of alternative system components might represent an option that is more cost-efficient in the long-term. Furthermore, the effect of experience curves for innovative technologies on possible cost decreases, should be taken into account to further enhance the accuracy of the analysis.

The economic assessment also needs to consider potential benefits. The cost analysis in this study shows that particularly for the case of Ethiopia, relatively high financial benefits can be expected, which can offset the total costs of sanitation, and even generate future profits. This shows that sanitation is not only a basic need, but that it can also be a viable business. For the case of Hamburg, benefits can be generated, but their monetary value depends on conditions such as the acceptance of products derived from human excreta. Provided that products derived from human excreta can be sold, it is possible that additional costs of recovery processes can be partly or fully recovered, as the analysis of System 3 NuRU (Hamburg) shows.

6.2.2 Integrating resource efficiency into planning and decision making

Decisions cannot be derived directly from the ceMFA model, but the results of the assessment can be used for decision support. One of the most important questions for decision makers will be whether the required efforts, particularly in terms of finances, justify a system change towards nutrient recovery. The model helps to quantify the required efforts and provides a basis for decision making.

In addition, the interrelation between the different resources is important, for example, the impact of nutrient recovery on the management of other resources such as water and energy. In this regard, the relative performance changes in urban sanitation, need to be considered within the framework of overall resource use. This means that the relevance of the studied criteria differs, and also that the contribution of the sanitation system with respect to the criteria differs. For example, in Germany the water and wastewater sector only contributes between 1% and 2% of the overall energy consumption. Although increasing energy efficiency in this sector is certainly important, it can only make a small contribution to overall energy savings. In contrast, up to about 30% of the current mineral phosphorus fertiliser use could be replaced by excreta-based phosphorus. This emphasises the need to further discuss the significance of the different criteria and to classify the possible contributions of the sanitation system. In general, it can be said that the focus should be on finite and essential resources such as phosphorus, which cannot be replaced by other resources, but for which sooner or later some kind of recycling will have to be initiated. This does not mean that recycling of nutrients should be prioritised at all costs, but the economical use of available resources is important and should always be a consideration. Resource efficiency criteria also need to be seen within a local context. For example, for the case of Ethiopia the availability of sustainable and affordable household fuels is at least as important as overall energy consumption.

As the discussion of transformation processes shows (sections 4.4 and 5.4), social aspects build a framework that inevitably needs to be considered if system changes in sanitation are to take place. This includes not only household behaviour and social acceptance, but also affordability. Even though these factors can be influenced, e.g. by pilot projects ("seeing is believing") and subsidies, they nevertheless play an important role in the decision environment. It should be kept in mind that water and sanitation do not only provide primary functions such as hygiene, environmental protection and resource recovery, but that they also fulfil socio-cultural functions. Local context is important in this regard.

Local conditions and priorities also impact on the drivers for system change (see sections 4.4.1 and 5.4.1). The list of drivers included in this study for both case studies should not be seen as an exhaustive list. Discussions show that both water and sanitation systems are subject to a variety of external factors, of which increased resource efficiency is only one ultimate objective. Decision makers will need to assess different water and sanitation systems when considering the development of drivers and objectives. This will require an integrated and holistic decision making process, for which assessments as carried out in this study will be helpful. In practice however, decision making in the water and sanitation sector often lacks holistic strategic planning; it is the result of a range of incremental (and often detached) planning steps[141]. For example, stricter treatment standards are usually complied with by additional treatment steps without questioning the functionality and appropriateness of the whole system. Also the relatively strong influence of households' decision making on urban systems is often underestimated[142]. There is therefore a need for organisations, institutions and individuals that understand the "big picture" and that are able to steer sanitation systems towards higher resource efficiency and sustainability. It is crucial that planning should not only include the interests of one process (e.g. the wastewater treatment plant), but should optimise the whole system.

[141] In many regions of the world, where sanitation is not given high priority, sanitation planning seems to be completely lacking and only relies on random implementation of facilities.

[142] A prominent example is the reduced household water consumption in Germany, which subsequently forces utilities to adapt accordingly, not only in terms of operation, but also in terms of cost recovery.

7 Conclusion

Optimising material flow management, which means that material flows should be controlled and managed in a manner that is environmentally and economically efficient, will become an important objective for decision makers of the cities of tomorrow. In urban water management and sanitation, the importance of improved water use, nutrient recovery and increased energy efficiency, is indisputable for improved resource efficiency. In addition, decision making needs to be based on the efficient use of economic resources.

The contribution of this research is twofold. Firstly, a method which is named ceMFA (cost, energy and material flow analysis) is developed to assess the resource efficiency of urban sanitation systems. Secondly, a comparative assessment of different systems aimed at nutrient recovery is carried out for two case studies, namely Hamburg in Germany and Arba Minch in Ethiopia.

The study shows that the method of ceMFA can provide a valuable tool for assessing the resource efficiency of different sanitation systems and can contribute comprehensively to systems analysis. A range of different sanitation systems, that show the potential to contribute to increased resource efficiency, is available for the two analysed case studies. Although none of the systems is superior in all criteria, the results represent a basis for decision making and further optimisation of the systems.

Introducing appropriate system transformations in sanitation can provide the opportunity to reduce the use of water, energy and finite resources such as phosphate rock. The ceMFA model allows the identification of such opportunities and highlights trade-offs and possible drawbacks. Using this work as a basis for decision support, it is now up to different stakeholders to consider resource efficiency in sanitation and to develop strategies for implementing more resource efficient systems.

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Annexes

List of Annexes

Annex A: Parameters of the Hamburg ceMFA model

Annex A summarises the data that was used as input parameters for the development of the ceMFA model for the city of Hamburg. The data is listed according to the processes as they are described in section 5.2. Average values as well as the standard deviation (abbreviated by stdev) are shown.

For the Monte Carlo simulations, the distribution of the parameter values can be selected to show different distributions, namely normal, tnormal (i.e. truncated normal), lognormal, tlognormal (i.e. truncated lognormal), and uniform. If a truncated distribution is selected, the minium and maximum values are included.

A.1 Process Agriculture

Parameter table for the Process Agriculture

Parameter	Unit	Average	Stdev	Distribution	Source
Agricultural area of Hamburg	ha	**19,189**	1919	normal	1)
Food uptake per person and year	kg p^{-1} y^{-1}	**726.3**	145.3	normal	Result of calculation based on 2)
Area needed to produce 1 kg of food	m^2 kg^{-1}	**4**	0.8	lognormal	3)
Evapotranspiration coefficient	-	**0.59**	0.05	tlognormal (0.2-1)	Estimation based on 4)
Non-central water supply to agriculture (Hamburg)	mio m^3 y^{-1}	**0.716**	0.05	normal	5)
Runoff coefficient on agricultural soil	-	**0.1**	0.05	tlognormal (0-0.5)	6)
Erosion in Hamburg	mio m^3 y^{-1}	**0.163**	0.0163	normal	7)
N losses of mineral fertiliser application	-	**0.10**	0.05	tlognormal (0-1)	8)
N content in food from agriculture in Germany	kg_N p^{-1} y^{-1}	**5.156**	1.547	tnormal (1-10)	Result of calculation based on 2)
N content in total food (incl. imports)	kg_N p^{-1} y^{-1}	**7.437**	2.231	normal	Result of calculation based on 2)
N in erosion and runoff of agricultural area of Hamburg	t_N y^{-1}	**163.13**	32.63	normal	7)
N in infiltration from agricultural area of Hamburg	t_N y^{-1}	**235.00**	47.0	normal	7)
N fertiliser demand	kg_N ha^{-1} y^{-1}	**94**	9.4	normal	9) & 18)

Parameter	Unit	Average	Stdev	Distribution	Source
N loss during urine fertilisation (gaseous losses)	-	**0.07**	0.05	lognormal	10)
P content in food from agriculture in Germany	$kg_P\ p^{-1}\ y^{-1}$	**0.739**	0.222	normal	Result of calculation based on 2)
P content in total food (incl. imports)	$kg_P\ p^{-1}\ y^{-1}$	**1.046**	0.314	normal	Result of calculation based on 2)
P in erosion and runoff in agricultural area of Hamburg	$t_P\ y^{-1}$	**2.571**	0.514	normal	7)
P in infiltration from agricultural area of Hamburg	$t_P\ y^{-1}$	**2.125**	0.425	normal	7)
P in atmospheric deposition	$kg_P\ ha^{-1}\ y^{-1}$	**0.37**	0.074	lognormal	7)
P fertiliser demand	$kg_P\ ha^{-1}\ y^{-1}$	**7.0**	1.0	normal	9) & 18)
C content in food from agriculture in Germany	$kg_C\ p^{-1}\ y^{-1}$	**74.258**	22.28	normal	Result of calculation based on 2)
C content in total food (incl. imports)	$kg_C\ p^{-1}\ y^{-1}$	**110.437**	33.13	normal	Result of calculation based on 2)
C in erosion and runoff in agricultural area of Hamburg	$t_C\ y^{-1}$	**1,957.51**	195.8	normal	Based on 7) & 11)
C in infiltration from agricultural area of Hamburg	$t_C\ y^{-1}$	**2,820.00**	282.0	normal	Based on 7) & 11)
Primary energy consumption for K production	$kWh\ kg_K^{-1}$	**2.42**	0.24	normal	12)
Primary energy consumption for N production	$kWh\ kg_N^{-1}$	**13.64**	1.36	normal	12)
Primary energy consumption for P production	$kWh\ kg_P^{-1}$	**2.14**	0.21	normal	12)
Primary energy producing ammonium sulfate	$kWh\ kg_S^{-1}$	**0**	0	uniform	12)
K content in food from agriculture in Germany	$kg_K\ p^{-1}\ y^{-1}$	**1.514**	0.454	normal	Result of calculation based on 2)
K content in total food (incl. imports)	$kg_K\ p^{-1}\ y^{-1}$	**2.58**	0.775	normal	Result of calculation based on 2)
K in erosion and runoff in agricultural area of Hamburg	$t_K\ y^{-1}$	**74.79**	22.44	normal	Based on 7) & 13)
K in infiltration from agricultural area of Hamburg	$t_K\ y^{-1}$	**61.82**	18.55	normal	Based on 7) & 13)
K in atmospheric deposition	$kg_K\ ha^{-1}\ y^{-1}$	**5.00**	1.000	lognormal	14)
K fertiliser demand	$kg_K\ ha^{-1}\ y^{-1}$	**21**	2.0	normal	9) & 18)

Parameter	Unit	Average	Stdev	Distribution	Source
S content in food from agriculture in Germany	$kg_S\ p^{-1}\ y^{-1}$	**0.394**	0.118	normal	Result of calculation based on 2)
S content in total food(incl. Imports)	$kg_S\ p^{-1}\ y^{-1}$	**0.597**	0.179	normal	Result of calculation based on 2)
S in erosion and runoff in agricultural area of Hamburg	$t_S\ y^{-1}$	**21.21**	6.362	normal	Based on 7) & 15)
S in infiltration from agricultural area of Hamburg	$t_S\ y^{-1}$	**30.55**	9.165	normal	Based on 7) & 15)
S in atmospheric deposition	$kg_S\ ha^{-1}\ y^{-1}$	**4.00**	0.800	normal	16)
S fertiliser demand	$kg_S\ ha^{-1}\ y^{-1}$	**20.00**	8.000	normal	17)

Sources: 1) Statistische Ämter des Bundes und der Länder, 2009, 2) FAO, 2004, 3) Baccini and Brunner, 1991, 4) Herrmann et al., 1997, 5) Statistisches Amt für Hamburg und Schleswig-Holstein, 2004, 6) DWA, 2005, 7) Behrendt et al., 2003, 8) Stroh and Djeradi, 2007, 9) Statistisches Bundesamt, 2007, 10) Kirchmann and Petterson, 1995, 11) Alberta Agriculture, 2008, 12) Patyk and Reinhardt, 1997, 13) Blume, 2004, 14) Blume et al., 1996, 15) Hassett and Banwart, 1992, 16) Smidt, 2007, 17) Kemira GrowHow GmbH., 2006, 18) IFA, 2008

A.2 Processes Surface Areas and Sewerage

Parameter table for the Processes Surface Areas and Sewerage

Parameter	Unit	Average	Stdev	Distribution	Source
Total area of Hamburg	ha	**75,500**	1,000	normal	1)
Area of surface waters in Hamburg	ha	**6,000**	300	normal	1)
Sewered area in Hamburg	ha	**29,500**	2,950	normal	2)
Area in Hamburg with separate sewer system	ha	**20,000**	2,000	normal	2)
Unsewered area in Hamburg	ha	**9,800**	980	normal	Calculated based on 1) & 2)
Infiltration coefficient on urban areas	-	**0.3**	0.05	tlognormal (0.05-0.8)	Estimated based on 3)
Rainfall in Hamburg	$m\ y^{-1}$	**0.77**	0.04	normal	4)
Evapotranspiration coefficient	-	**0.59**	0.05	tlognormal (0.2-1)	Estimated based on 3)
Combined sewer overflow in Hamburg (% of total flow)	-	**0.03**	0.01	tlognormal (0-0.1)	2)
Infiltration rate into sewer	$m^3\ ha^{-1}_{sewer\ area}\ y^{-1}$	**700**	40	normal	Calculated based on 2)
Exfiltration from sewer (related to total volume)	-	**0.023**	0.002	tlognormal (0-0.1)	Calculated based on 2)

Parameter	Unit	Average	Stdev	Distribution	Source
Calculated runoff coefficient on urban areas (includes ratio rainfall to sewer)	-	**0.216**	0.022	tlognormal (0.01-0.8)	Calculated based on 2)
N in runoff of surface areas	$g_N\ l^{-1}$	**0.0025**	0.001	lognormal	5), 14)
P in runoff	$g_P\ l^{-1}$	**0.0003**	0.0001	lognormal	6), 14)
TOC in runoff	$g_C\ l^{-1}$	**0.05**	0.03	lognormal	7), 14)
K in runoff	$g_K\ l^{-1}$	**0.0014**	0.0018	lognormal	6) & 8)
S in runoff	$g_S\ l^{-1}$	**0.0016**	0.0026	lognormal	6)
N in CSO as % of total wastewater input	-	**0.01**	0.002	lognormal	2)
P in CSO as % of total wastewater input	-	**0.01**	0.002	lognormal	2)
K in CSO as % of total wastewater input	-	**0.01**	0.003	lognormal	Assumption
S in CSO as % of total wastewater input	-	**0.01**	0.003	lognormal	Assumption
N in atmospheric deposition	$kg_N\ ha^{-1}\ y^{-1}$	**8.0**	5	lognormal	9) & 10)
P in atmospheric deposition	$kg_P\ ha^{-1}\ y^{-1}$	**0.37**	0.074	lognormal	10) & 11)
TOC in atmospheric deposition (in rainfall)	$g_C\ l^{-1}$	**0.005**	0.001	lognormal	6)
S in atmospheric deposition	$kg_S\ ha^{-1}\ y^{-1}$	**4.0**	2.0	lognormal	9), 12)
K in atmospheric deposition	$kg_K\ ha^{-1}\ y^{-1}$	**5.0**	1.000	lognormal	12)
Electricity consumption conventional sewer	$kWh_{el}\ m^{-3}$	**0.12**	0.012	lognormal	Calculated based on 13)

Sources: 1) Statistische Ämter des Bundes und der Länder, 2009, 2) Hamburger Stadtentwässerung (HSE), 2000, 3) Herrmann et al., 1997, 4) BSU, 2005, 5) Herrmann and Klaus, 1997, 6) Dierkes et al., 2005, 7) Hahn, 2010, 8) Santaularia Lietz, 2001, 9) Smidt, 2007, 10) Tidaker et al., 2007, 11) Behrendt et al., 2003, 12) Blume et al., 1996, 13) Balkema, 2003, 14) Peters, 2007

A.3 Process Composting

Parameter table for the Process Composting

Parameter	Unit	Average	Stdev	Distribution	Source
Compost per ton biowaste & faeces	$t\ t^{-1}_{orgwaste}$	**0.73**	0.073	tnormal (0.5-1)	Calculated based on 1)
Electricity consumption for centralised composting process	$kWh_{el}\ t^{-1}_{input}$	**44**	5.000	lognormal	2), 7)
N transfer coefficient to compost (organic waste)	-	**0.750**	0.225	tnormal (0.2-1)	3), 5), 6)
N transfer coefficient to compost (organic waste & faeces)	-	**0.650**	0.195	tnormal (0.2-1)	3)
P transfer coefficient to compost (organic waste & faeces)	-	**0.99**	0.1	tnormal (0.5-1)	6)
TOC transfer coefficient to compost (organic waste & faeces)	-	**0.45**	0.135	tnormal (0-1)	4), 6)

Parameter	Unit	Average	Stdev	Distribution	Source
K transfer coefficient to compost (organic waste & faeces)	-	**0.85**	0.255	tnormal (0-1)	Assumption
S transfer coefficient to compost (organic waste & faeces)	-	**0.45**	0.135	tnormal (0-1)	Assumption

Sources: 1) BSU, 2007, 2) Thomé-Kozmiensky, 1998, 3) Amlinger et al., 2005, 4) Belevi, 2002, 5) Sonesson et al., 1997, 6) Leitzinger, 1999, 7) Schmelz, 2000

A.4 Process Households

Parameter table for the Process Households

Parameter	Unit	Average	Stdev	Distribution	Source
Inhabitants of Hamburg	p	**1,741,182**	174,118	normal	1)
Number of households in Hamburg	hh	**883,045**	88,002	normal	2)
Specific water consumption	$l\ p^{-1}\ d^{-1}$	**107**	10	lognormal	3)
Water loss in households (cleaning, watering flowers…)	$l\ p^{-1}\ d^{-1}$	**4**	2	uniform (2.0-6.0)	4)
Greywater from households	$l\ p^{-1}\ d^{-1}$	**68**	7	tnormal (10-150)	5)
Drinking water per person & day	$l\ p^{-1}\ d^{-1}$	**5**	1	tnormal (2-7)	5)
Flushing water conventional toilet	$l\ p^{-1}\ d^{-1}$	**30**	10	tnormal (10-50)	Assumption
Flushing water vacuum toilet	$l\ p^{-1}\ d^{-1}$	**6**	1	tnormal (3-9)	Assumption
Water for urine flushing, sep. toilet	$l\ p^{-1}\ d^{-1}$	**0**		uniform	Assumption
Water for faeces flushing, sep. toilet	$l\ p^{-1}\ d^{-1}$	**3**	0.3	tnormal (0.5-14)	Assumption
Amount of N per kg TS in biowaste	$kg_N\ kg_{TS}^{-1}$	**0.018**	0.006	tnormal (0-0.3)	5)
N load in urine per person	$g_N\ p^{-1}\ d^{-1}$	**9.50**	2.60	tnormal (3.6-16)	5)
N load in faeces per person	$g_N\ p^{-1}\ d^{-1}$	**1.70**	0.700	tnormal (0.25-4.2)	5)
N load in greywater per person	$g_N\ p^{-1}\ d^{-1}$	**1.00**	0.400	tnormal (0.1-2.0)	5)
Amount of P per kg TS in biowaste	$kg_P\ kg_{TS}^{-1}$	**0.003**	0.001	tnormal (0.001-0.005)	5)
P load in urine per person	$g_P\ p^{-1}\ d^{-1}$	**1.0**	0.400	tnormal (0.4-2.5)	5)
P load in faeces per person	$g_P\ p^{-1}\ d^{-1}$	**0.600**	0.400	tnormal (0.3-1.7)	5)
P load in greywater per person	$g_P\ p^{-1}\ d^{-1}$	**0.500**	0.300	tnormal (0.1-2.2)	5)
Org. waste production per person	$kg\ p^{-1}\ y^{-1}$	**91.8**	40.1	normal	5)
Collection rate of biowaste	-	**0.22**	0.022	normal	6)
Detergents etc. per person	$kg\ p^{-1}\ y^{-1}$	**10**	1	normal	7)

Parameter	Unit	Average	Stdev	Distribution	Source
TS content of biowaste	-	**0.39**	0.163	tnormal (0.14-0.68)	5)
Urine volume per person	$l\ p^{-1}\ d^{-1}$	**1.27**	0.32	tnormal (0.5-2.5)	5)
Collection rate of urine	-	**0.70**	0.1	tnormal (0-1)	8)
Faeces per person	$kg\ p^{-1}\ d^{-1}$	**0.15**	0.04	tnormal (0.07-0.4)	5)
TS content of Blackwater	-	**0.011**	0.007	lognormal	5)
VS content of organic waste (total)	-	**0.27**	0.07	lognormal	5)
VS content of Blackwater	-	**0.008**	0.005	lognormal	5)
TOC in urine produced per person	$g_C\ p^{-1}\ d^{-1}$	**5.30**	3.100	tnormal (2.3-8.7)	5)
TOC in faeces produced per person	$g_C\ p^{-1}\ d^{-1}$	**26.20**	10.700	tnormal (17.7-46.6)	5)
TOC in greywater produced per person	$g_C\ p^{-1}\ d^{-1}$	**13.30**	2.000	tnormal (11.4-15.1)	5)
TOC per kg TS biowaste	$kg_C\ kg_{TS}^{-1}$	**0.44**	0.060	tnormal (0.37-0.53)	5)
Bottled drinking water production primary energy	$kWh\ l^{-1}$	**0.25**	0.05	lognormal	10)
K per kg TS biowaste	$kg_K\ kg_{TS}^{-1}$	**0.0077**	0.0022	tnormal (0.004-0.054)	5)
K load in urine per person	$g_K\ p^{-1}\ d^{-1}$	**2.40**	0.60	tnormal (1.0-3.8)	5)
K load in faeces per person	$g_K\ p^{-1}\ d^{-1}$	**0.70**	0.300	tnormal (0.30-1.10)	5)
K load in greywater per person	$g_K\ p^{-1}\ d^{-1}$	**1.30**	1.100	tnormal (0.00-4.10)	5)
Concentration of K in bottled water	$g_K\ l^{-1}$	**0.0111**	0.0096	lognormal	9)
S per kg TS biowaste	$kg_S\ kg_{TS}^{-1}$	**0.0023**	0.00020	tnormal (0.0020-0.0024)	5)
S in urine produced per person	$g_S\ p^{-1}\ d^{-1}$	**0.83**	0.300	tnormal (0.30-1.40)	5)
S in faeces produced per person	$g_S\ p^{-1}\ d^{-1}$	**0.20**	0.040	tnormal (0.07-0.4)	5)
S in greywater produced per person	$g_S\ p^{-1}\ d^{-1}$	**3.50**	2.70	tnormal (0.50-7.70)	5)
Concentration of S in bottled water	$gS\ l^{-1}$	**0.0115**	0.007	lognormal	9)

Sources: 1) Statistisches Amt für Hamburg und Schleswig-Holstein, 2008b, 2) Statistisches Amt für Hamburg und Schleswig-Holstein, 2008a, 3) Hamburg Wasser, 2007a, 4) Leonhardt, 2005, 5) Average and standard deviation of extensive literature review and analysis (see section 2.4.2 and Annex B), 6) BSU, 2007, 7) UBA, 2008, 8) Jönsson, 2001, 9) FineWaters, 2008, 10) Leist, 2002

Cost parameters related to the Process Household

Parameter	Unit cost	Unit	Life-span (years)	**Selected average & standard deviation**[1]	Distribution	Source
Conventional toilet	250-300	€	20	**275±25**	normal	1), 2)
Urine diversion toilet	450-950	€	20	**550±75**	normal	1), 2), 3)
Vacuum toilet	600-980	€	20	**650±75**	normal	1), 2), 3)
Urine diversion dry toilet	700-1500	€	20	**800±100**	normal	4)
Ventilation dry toilet	600	€	20	**600±60**	normal	2)
Vacuum pipe in-house	40-50	€ m^{-1}	50	**40±4**	normal	1), 2)
Urine pipe	15-40	€ m^{-1}	50	**20±2**	normal	1), 2)
Retrofitting into existing buildings	948-2347	€ hh^{-1}	50	**1,500±200**	normal	8)
Sewer house connection	920	€	50	**900±50**	normal	1)
Vacuum house connection	300-400	€	50	**300±50**	normal	1), 5)
Urine house connection	300	€	50	**300±50**	normal	Estimation
Urine storage tank[2]	302-960	€ m^{-3}	50	**500±40**	normal	Based on 1), 6), 7), 8)
Maintenance urine pipes & composting toilet	-	% of invest	-	**3.0±0.3**	normal	Based on 1), 2), 8)
Maintenance pipes, house connections & urine storage	-	% of invest	-	**1.0±0.1**	normal	Estimation
Operation composting toilet	15-20	€ p^{-1} y^{-1}	-	**15±3**	normal	2), 9)

Sources: 1) Oldenburg and Dlabacs, 2007; 2) Herbst, 2008; 3) Balkema, 2003; 4) Werner et al., 2006; 5) Halbach, 2003; 6) Görges, 2008; 7)Drücker, 2004; 8) Starkl et al., 2005; 9) Kaufmann et al., 2007

[1] Please note that for toilet types that are not yet widely used the average is selected at the lower range of the unit costs. This is due to the fact that a large scale implementation will result in a decrease of unit costs (economies of scale).

[2] It is assumed that the cost for on-site urine storage includes costs for pipes outside the building.

A.5 Process Centralised Water Supply

Nutrient concentrations in drinking water

	Unit	N	P	S	K	Source
Groundwater	mg l^{-1}	1±1	-	no data[3]	no data[3]	Based on Kröning, 2008
Drinking water	mg l^{-1}	0.43±0.05	-	1.7±1.0	2.5±0.8	Calculated from Hamburg Wasser, 2007b

- *not applicable, for all parameters lognormal distributions are assumed*

A.6 Process Wastewater Treatment Plant

Required effluent concentrations (AbwV, 2004)

Parameter	Value	Unit
COD[4]	75	mg l^{-1}
N	13	mg l^{-1}
P	1	mg l^{-1}

Parameter table for the Processes Wastewater Treatment Plant and Sludge Treatment

Parameter	Unit	Average	Stdev	Distribution	Source
Screenings, sand etc. per person	kg p^{-1} y^{-1}	**3.9**	0.39	normal	1)
Amount of primary sludge	g_{TS} p^{-1} d^{-1}	**10**	1	normal	2)
Sludge per eliminated COD	kg_{VSS} kg_{COD}^{-1}	**0.4**	0.04	tnormal (0-1)	3) & 10)
Sludge per eliminated N	kg_{TS} kg_{N}^{-1}	**1.059**	0.1059	normal	calculated based on 4)
Sludge per eliminated P	kg_{TS} kg_{P}^{-1}	**6.8**	0.68	normal	5)
Degree of TOC removed in the primary sedimentation	-	**0.3**	0.03	tnormal (0-0.6)	5)
TOC transfer coefficient to sludge liquor	-	**0.07**	0.007	lognormal	6)
Maximum concentration of COD in the effluent of WWT	mg_{COD} l^{-1}	**75**	0	uniform	7)

3 A 90% removal of S and K in groundwater treatment is assumed for the analysis

4 In the influent, a ratio of COD/TOC=3±0.4 is assumed (Wentzel et al., 2003), whereas in the effluent a ratio of COD/TOC=4±0.7 is assumed (Gulyas, 2003).

Parameter	**Unit**	**Average**	**Stdev**	**Distribution**	**Source**
TOC transfer coefficient anaerobic digestion sewage sludge to biogas	-	**0.45**	0.05	tnormal (0.2-0.7)	based on 8)
TOC transfer coefficient anaerobic digestion blackwater to biogas	-	**0.6**	0.06	tnormal (0.1-0.9)	8)
Ratio of TOC to COD in the effluent of the WWTP	-	**0.25**	0.05	tnormal (0.15-0.35)	9)
Ratio of COD to TOC in the wastewater (influent)	-	**3**	0.4	tnormal (2-4)	10)
TOC transfer coefficient to sludge	-	**0.67**	0.1	tnormal (0.5-0.8)	5), 10)
Degree of N removed as primary sludge	-	**0.09**	0.01	lognormal	5)
Degree of N incorporated into biomass per COD removed	% of COD load	**2.25**	0.45	lognormal	5)
N transfer coefficient to sludge liquor	-	**0.83**	0.166	tnormal (0.5-1)	6)
Efficiency of N removal in nitrification & denitrification	-	**0.8**	0.05	tnormal (0.7-1)	Assumption
Maximum concentration of total N in the effluent of WWT	$mg_N\ l^{-1}$	**13**	0	uniform	7)
Degree of P removed in primary sludge	-	**0.08**	0.01	lognormal	5)
Incorporation of P in biomass per COD removed	% of COD load	**0.5**	0.2	lognormal	5)
P transfer coefficient to sludge liquor	-	**0.05**	0.005	lognormal	6)
Maximum concentration of total P in the effluent of WWT	$mg_P\ l^{-1}$	**1**	0	uniform	7)
Degree of K removed as primary sludge	-	**0.1**	0.01	lognormal	Assumption
Degree of K incorporated into biomass per COD	% of COD load	**0.18**	0.018	lognormal	12)
K transfer coefficient to sludge liquor	-	**0.77**	0.231	tnormal (0-1)	11)
Degree of S removed as primary sludge	-	**0.1**	0.01	lognormal	Assumption
Degree of S incorporated into biomass per COD	% of COD load	**0.18**	0.036	lognormal	12)
S transfer coefficient to sludge liquor	-	**0.2**	0.06	tnormal (0-1)	Assumption
S transfer coefficient sludge incineration to ash	-	**0.6**	0.12	tnormal (0-1)	13)
S transfer coefficient anaerobic digestion to biogas	-	**0.6**	0.18	tnormal (0-1)	Based on 8)
TS content of sludge before sludge treatment	-	**0.03**	0.003	lognormal	1)
TS content of sludge before incineration	-	**0.42**	0.042	normal	1)
VS/TS ratio of sewage sludge	-	**0.65**	0.05	normal	14)
TS content of digested sludge	-	**0.033**	0.0033	lognormal	15)

Parameter	Unit	Average	Stdev	Distribution	Source
VS content of sludge before sludge treatment	-	**0.02**	0.002	lognormal	Assumption
Biogas production per kg VS in sludge before sludge treatment	m^3 BG kg_{VS}^{-1}	**0.5**	0.05	normal	16)
Incineration residues as % of sludge before incineration	-	**0.2**	0.02	lognormal	1)
Total solids after dewatering (TS ratio)	-	**0.22**	0.022	tnormal (0-0.5)	15)
Total solids after thickening (TS ratio)	-	**0.05**	0.005	tnormal (0-0.1)	15)
Total solids after incineration (TS ratio)	-	**0.99**	0.099	tnormal (0-1)	15)

Sources: 1) Hamburger Stadtentwässerung (HSE), 2000, 2) Thomé-Kozmiensky, 1998, 3) Tchobanoglous, 2003, 4) Halling-Sorensen and Jorgensen, 1993, 5) ATV-DVWK, 2000, 6) pers. communication Thomas Werner (HSE), 7) AbwV, 2004, 8) Wendland, 2008, 9) Gulyas, 2003, 10) Wentzel et al., 2003, 11) Jardin and Pöpel, 1994, 12) Estimation based on ratio K/N and S/N in bacteria=0.08 see 3), 13) Baccini and Brunner, 1991, 14) Wendler, 2005, 15) Thierbach and Hanssen, 2002, 16) Müller et al., 1999, 17) Arlt, 2003

Transfer coefficients sludge (see also table above)

Parameter	Average	Stdev	Unit
Transfer coefficients settling:			
TOC	0.3	0.03	-
N	0.09	0.01	-
P	0.08	0.01	-
K	0.1	0.01	-
S	0.1	0.01	-
Transfer coefficients excess sludge (biomass):			
TOC	0.67	0.1	-
N	0.023	0.005	of COD load in excess sludge
P[5]	0.005	0.002	
K	0.002	0.0002	
S	0.002	0.0002	
Transfer coefficients sludge liquor:			
TOC	0.07	0.007	-

[5] Please note that this value represents only the biological incorporation. In addition, there is P in sludge from chemical precipitation. This amount is calculated separately based on the assumption that the required discharge standards are fully met.

Parameter	Average	Stdev	Unit
N	0.83	0.17	-
P	0.05	0.005	-
K	0.77	0.23	-
S	0.2	0.06	-

The values are based on ATV-DVWK, 2000, Remy and Ruhland, 2006, Wentzel et al., 2003, Tchobanoglous, 2003 and own assumptions. The transfer coefficients for the calculation of substance flows in sludge liquor are based on personal information from the Hamburg wastewater treatment plant (Thomas Werner).

Dry matter contents and volume calculations

The recalculation of sludge volumes after treatment processes (i.e. after the extraction of water) is based on following formula:

$V_1 / V_2 = TS_2 / TS_1$

with:
V_1: initial volume
V_2: final volume
TS_1: initial dry matter content
TS_2: final dry matter content

Dry matter contents used for the sludge treatment modelling (inflow to the respective processes)

	Thickening TS [%] / VS [%]	Digestion TS [%] / VS [%]	Dewatering TS [%]	Drying TS [%]	Incineration TS [%]
Sewage sludge	3[a] / 2[d]	5[b] / 3.25[d]	3.3[b]	22[b]	42[b]
Organic waste[6]:		39[c] / 27[c]			
Blackwater[7]		1.1[c]/ 0.8[c]			

Sources: [a] Hamburger Stadtentwässerung (HSE), 2000, [b] Thierbach and Hanssen, 2002, [c] Literature review (see Section 2 and Annex B), [d] Assumption: 65% of TS

[6] For systems NuRU & CoDig

[7] For system CoDig

Energy consumption parameters for centralised treatment of wastewater and sludge

Parameter	Unit	Average	Stdev	Distribution	Source
Primary treatment, pumping, recirculation etc.	$kWh_{el}\ m^{-3}{}_{wastewater}$	**0.4**	0.15	lognormal	1) & 2)
WWTP infrastructure	$kWh_{el}\ y^{-1}$	**158,000**	30,000	normal	Based on 2)
Aeration (COD removal)	$kWh_{el}\ kg^{-1}{}_{COD\ elim}$	**0.35**	0.05	lognormal	Based on 5) & 9)
Nitrification[8]	$kWh_{el}\ kg^{-1}{}_{NH4\text{-}N\ elim}$	**3.8**	0.7	lognormal	3), 4) & 5)
Phosphorus precipitation[9]	$kWh_{el}\ kg^{-1}{}_{P}$	**5**	2.2	tnormal (1.4-8.6)	4, 5) & 6)
Sludge thickening	$kWh_{el}\ m^{-3}{}_{sludge}$ entering thickening	**0.5**	0.2	lognormal	2)
Sludge digestion:					
Sludge heating	$kWh_{heat}\ t^{-1}{}_{sludge}$	**26.7**	0	uniform	Calculation, see text
Heat losses	mio $kWh_{heat}\ y^{-1}$	**2.19**	0	uniform	Calculation, see text
Sludge dewatering	$kWh_{el}\ t^{-1}{}_{TS\ sludge}$	**55**	15	tnormal (30-80)	2) & 7)
Sludge drying	$kWh_{PE}\ t^{-1}{}_{TS\ sludge}$	**2,500**	500	tnormal (1,500-3,500)	7)[10]
Sludge incineration	m^3 nat. gas $t^{-1}{}_{sludge,\ 42\%TS}$	**20**	2	tnormal (10-30)	8)

Sources: 1) Tchobanoglous, 2003, 2) 1999, 3) Dockhorn and Dichtl, 2006, 4) Maurer et al., 2003, 5) Karlsson, 1996, 6) Levlin and Hultman, 2003, 7) Arlt, 2003, 8) Mininni et al., 1997, 9) Dockhorn, 2006

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[8] Data on energy requirements for nitrogen elimination in literature sometimes include also the provision of external carbon sources such as methanol for denitrification (Maurer et al., 2003; Wilsenach, 2006). This is not considered here. The impact of this heterotrophic process on the energy consumption for removal of organic matter (reduced external oxygen requirements) or energy production (reduced generation of biogas) is neglected. Energy requirements for recirculation are included in the calculation dependent on total water flows.

[9] Maurer et al. (2003) state a primary energy demand of 24MJ $kg^{-1}{}_{P}$ for precipitant production ($FeSO_4$). Using the European electricity mix efficiency of 0.31 cited by them, this results in an electricity consumption of 7.44 kWh_{el} $Kg^{-1}{}_{P}$. They add also other energy consuming processes such as sludge handling and incineration to the energy demand for P removal. However, these are not included here within the parameter ‘P precipitation’, since the energy demand for sludge treatment is calculated separately based on mass flows including the consideration of additional sludge volume due to precipitation. Karlsson (1996) and Levlin and Hultman (2003) state an energy consumption for chemicals used for precipitation ranging between 20-45 Wh m^{-3}. Assuming an average P concentration in mixed wastewater of 9 g m^{-3} (ATV-DVWK, 2000), this results in an electricity demand of 2.2-5 kWh_{el} $Kg^{-1}{}_{P}$. For dosing, Müller et al. (1999) cite an electricity demand of 0.5Wh m^{-3} wastewater, which can be converted to approximately 0.06 kWh $kg^{-1}{}_{P}$. Yet, this value is neglected in the overall balance.

[10] Drying of organic waste needs about 67% of the energy consumption of sewage sludge (Arlt, 2003).

A.7 Process Transport

Stations for secondary processing

Name of station	number of districts	number of inhabitants
Bergedorf	16	224,227
Harburg	23	218,160
West	41	777,736
North	21	496,645

The following table summarises the results of the mileage calculation including the general points of origin and destination of the different source-separated flows. As mentioned above trips 1 to 5 are derived from the GIS based model. Trips 6 to 10 are determined by calculating the total mass flow of the respective flows in one year divided by the maximum capacity of one lorry. This procedure results in the number of required trips per year. This number is subsequently multiplied by an assumed distance of 25 km times two (i.e. return) to arrive at the total required mileage.

Required lorry-based transports for fortnightly collections

Trip No.	Flow	Origin	Destination	System	Mileage per year [km y^{-1}]
1	Urine	Household/ quarter	Treatment or storage (4 stations)	3 NuRU 6 CompU	693,879
2	Blackwater	Temporary storage	Centralised treatment	4 CoDig	11,084,050
3	Blackwater & organic waste	Household/ quarter	Storage (4 stations)	5 BlaD	5,071,874
4	Organic waste	Household	Centralised treatment	1 CurS 2 NuRS 3 NuRU 4 CoDig 6 CompU	251,278
5	Faeces	Household	Composting plant	6 CompU	444,824
6	Urine products (MAP, NH_4)	Treatment in stations	Agriculture	3 NuRU	219,500
7	Urine	Storage in stations	Agriculture	6 CompU	1,765,584
8	Sludge products (MAP etc.)	Centralised treatment	Agriculture	2 NuRS 4 CoDig	77,595 317,936
9	Digested Slurry	Storage in stations	Agriculture	5 BlaD	14,855,134
10	Compost	Centralised treatment	Agriculture	1 CurS 2 NurS 6 CompU	126,091 126,091 606041

Energy parameters for the Process Transport

Parameter	Unit	Average & Stdev	Distribution
Fuel consumption, liquids transport, full	$l\ km^{-1}$	0.35±0.04	uniform (0.30-0.40)
Fuel consumption, liquids transport, empty	$l\ km^{-1}$	0.29±0.03	uniform (0.25-0.33)
Fuel consumption, solids transport, full	$l\ km^{-1}$	0.24±0.02	uniform (0.20-0.28)
Fuel consumption, solids transport, empty	$l\ km^{-1}$	0.18±0.02	uniform (0.15-0.21)
Lower heating value fuel	$kWh\ l^{-1}$	8.9±0.3	normal
Primary energy factor fuel	-	1.1±0.1	tnormal (1-2)

Data based on IFEU and SGKV, 2002; ORNL, 2008; DIN, 2007

A.8 Process Nutrient Recovery from Sewage Sludge

Parameters used to calculate the flows of the Process Nutrient Recovery from Sewage Sludge

Parameter	Unit	Average & Stdev	Distribution	Source
Mass flow$_{inputs}$				
MgO	$t_{MgO}\ t_{P\ product}^{-1}$	1.3*1.5	tnormal (1-3)	molar ratio & hyperstochiometric dosing
H_2SO_4	$t_{H2SO4}\ t_{S\ product}^{-1}$	3.06	tnormal	molar ratio[11]
NaOH	$kg\ m^{-3}$	0.9±0.2	(0.5-1.2)	Esemen and Dockhorn, 2009
Mass flow$_{product}$	$t\ t_{TS\ input}^{-1}$	0.31±0.03	normal	Seaborne EPM AG, 2003
Carbon$_{product}$	t	0	uniform	Assumption
Nitrogen$_{sludge\ liquor}$[12]	$mg\ l^{-1}$	50±5	normal	Bayerle, 2007 cited in Montag, 2008
Phosphorus$_{product}$	$t_P\ t_{Pinput}^{-1}$	0.65±0.05	tnormal (0.2-0.9)	Montag, 2008
Potassium$_{product}$	-	0	uniform	Assumption
Sulphur$_{product}$[13]	$t_S\ t_{Ni\ n\ NH3\ solution}^{-1}$	1.2±0.1	normal	Based on Dockhorn, 2007

11 The sulphur in the product is calculated depending on the nitrogen to be stripped

12 The nitrogen in the product is calculated based on the condition that the nitrogen concentration in the sludge liquor is 50mgNH$_4$-N (Bayerle, 2007 cited in Montag, 2008) and that the remaining nitrogen is fully recovered in the products either as MAP or as $(NH_4)_2SO_4$

13 Sulphuric acid is used to recover the ammonia in the scrubber. The amount of S needed is calculated based on the total amount of N in the products of the recovery process minus the N incorporated in MAP.

Parameters used to calculate the energy demand of the process nutrient recovery from sludge

Parameter	Unit	Average & Stdev	Distribution	Source
MgO production	kWh kg_{Mg}^{-1}	3.7±0.2	normal	Based on Patyk and Reinhardt, 1997
MAP precipitation	kWh_{el} $m^{-3}_{wastewater}$	1.6±0.1	lognormal	UBA and Öko-Institut, 2000
Stripping				
Heat	kWh_{th} $m^{-3}_{wastewater}$	42.5±2.5	normal	Haberkern et al., 2008
Electricity	kWh_{el} $m^{-3}_{wastewater}$	3.15±0.35	normal	Haberkern et al., 2008
NaOH production	kWh_{el} $tNaOH^{-1}$	1,200±135	normal	UBA and Öko-Institut, 2000
H_2SO_4 production[14]		-		neglected

Assumptions for the AshDec process (system modification in Section 4.3.3, not used for modelling)

Parameter	Unit	Average	Source
P recovery rate	$t_{P\ fertiliser}$ $t_{P\ ash}^{-1}$	> 0.90	AshDec, 2009
P content fertiliser	t_{P2O5} $t_{fertiliser}$	0.12	Hermann, 2008
K content fertiliser	t_{K2O} $t_{fertiliser}$	0.20	Hermann, 2008
Total production cost	€ $t_{fertiliser}$	170-270	Ante, 2009
Electricity consumption	kWh_{el} $t_{fertiliser}$	200-300	calculated based on Ante, 2009

Unit costs of additives in nutrient recovery processes

Additive	Unit cost [€ t^{-1}]	Source	Selected average and stdev [€ t^{-1}]	Distribution
NaOH	200-430	1, 2	300±60	normal
H_2SO_4 (96%)	100-297	2, 3	200±40	normal
MgO	280-406	1, 3, 4	300±30	normal

Sources: 1) Pinnekamp et al., 2007, 2) Esemen and Dockhorn, 2009, 3) Ek et al., 2006, 4) Dockhorn, 2007

[14] Patyk and Reinhardt (1997) cite a negative energy value for sulphuric acid as it is a product of energy conversion processes. However, this is not considered here.

A.9 Processes Nutrient Recovery from Sludge Liquor and Sludge Ash

Parameters used to calculate the output flows to agriculture of the Processes Nutrient Recovery from Sludge Liquor and Phosphorus Recovery from Ash

Parameter	Unit	Average & Stdev	Distribution	Source
Mass flow$_{inputs}$				
Stripping				
H_2SO_4	$t_{H2SO4}\ t_{N\ eliminated}^{-1}$	3.8±0.1	normal	Haberkern et al., 2008
NaOH	$t_{NaOH}\ t_{N\ eliminated}^{-1}$	4.0±0.5	normal	Haberkern et al., 2008
BioCon				
H_2SO_4	$t_{H2SO4}\ t_{TS\ input}^{-1}$	0.287±0.030	lognormal	Pettersson, 2001
HCl	$t_{HCl}\ t_{TS\ input}^{-1}$	0.063±0.006	lognormal	Pettersson, 2001
KCl	$t_{KCl}\ t_{TS\ input}^{-1}$	0.116±0.012	lognormal	Pettersson, 2001
NaOH	$t_{NaOH}\ t_{TS\ input}^{-1}$	0.010±0.001	lognormal	Pettersson, 2001
Mass flow$_{product}$				
H_3PO_4	$t\ t_{TS\ input}^{-1}$	0.066±0.007	lognormal	Pettersson, 2001
$(NH_4)_2SO_4$ (41%)	$t\ t_{N\ eliminated}^{-1}$	12±0.5	normal	Haberkern et al., 2008; Dockhorn, 2007
Carbon$_{product}$	t	0	uniform	Assumption
Nitrogen$_{product}$ $(NH_4)_2SO_4$	$t_N\ t_{N\ input}^{-1}$	0.90±0.05	tnormal (0.4-1.0)	{Bosshart, 1993 714 /id}
Phosphorus$_{product}$ H_3PO_4	$t_P\ t_{P\ input}^{-1}$	0.61±0.06	tnormal (0.2-1.0)	Balmer, 2004
Potassium$_{product}$	-	0	uniform	Assumption
Sulphur$_{product}$[15]	$t_S\ t_{N\ eliminated}^{-1}$	1.2±0.1	tnormal (0-2)	Based on Dockhorn, 2007

A.10 Process Nutrient Recovery from Urine

Parameters used to calculate the output flows to agriculture of the Process Nutrient Recovery from Urine

Parameter	Unit	Average & Stdev	Distribution	Source
Mass flow (MAP & NH_3)				
MAP	$t\ t_{Pinput}^{-1}$	8±0.2	normal	Calculated based on molar masses (12.6% P in MAP)
NH_3 solution (41% $(NH_4)_2SO_4$)	$t\ t_{Neliminated}^{-1}$	12±0.5	normal	Haberkern et al., 2008; Dockhorn, 2007
Carbon	$t\ t_{substrate}^{-1}$	0	uniform	Assumption
Nitrogen[16]				
MAP	$t_N\ t_{P\ in\ MAP}^{-1}$	0.45±0.01	lognormal	Based on molar masses N and P
$(NH_4)_2SO_4$ solution	$t_N\ t_{N\ after\ MAP}^{-1}$	0.94±0.04	tnormal (0.4-1.0)	Tettenborn et al., 2007; Dockhorn, 2007

[15] Sulphuric acid is used to recover the ammonia in the scrubber. The amount of S needed is calculated based on the total amount of N transferred from the sludge liquor to the product.

Parameter	Unit	Average & Stdev	Distribution	Source
Phosphorus	$t_P\ t_{P\ input}^{-1}$	0.97±0.02	tnormal (0.5-1.0)	Maurer et al., 2006; Lind et al., 2000
Potassium	-	0	uniform	Assumption based on Udert et al., 2004
Sulphur[17]	$t_S\ t_{N\ in\ NH3\ solution}^{-1}$	1.2±0.1	tnormal (1.0-2.0)	Based on Dockhorn, 2007

Parameters used to calculate the energy demand of the process nutrient recovery from urine

Parameter	Unit	Average & Stdev	Distribution	Source
Molar ratio of dosing MgO:P	$tMg\ tP_{input}^{-1}$	1.5±0.2	tnormal (1.0-2.5)	Based on Gethke et al., 2006; Li et al., 2007
Energy for MgO production	$kWh\ kgMg^{-1}$	3.7±0.2	normal	Based on Patyk and Reinhardt, 1997
Energy for MAP process	$kWh\ kgN_{in\ MAP}^{-1}$	1.6±0.2	lognormal	Siegrist, 1996, cited in Maurer et al., 2003
Energy for stripping	$kWh\ m^3_{urine}{}^{-1}$	70±14	tnormal (42-97)	Assumption based on Tettenborn et al., 2007; Maurer et al., 2003
Energy for H_2SO_4 production[18]	-	-	-	Neglected

[16] The nitrogen in the output is calculated as sum of N in MAP and in NH_3 solution. First, the nitrogen load in MAP is calculated dependent on the phosphorus load in MAP with a molar ratio of 1:1. This N load is then deducted from the input, and the transfer coefficient (=efficiency) for steam stripping is applied.

[17] The sulphur in the output derives from sulphuric acid added to the process to bind the stripped NH_3 as ammonium sulphate.

[18] Patyk and Reinhardt (1997) cite a negative energy value for sulphuric acid as it is a product of energy conversion processes. However, this is not considered here.

A.11 Anaerobic Digestion of Blackwater

Parameters used to calculate the energy production of the Process Anaerobic Digestion of Blackwater

Parameter	Unit	Average & Stdev	Distribution	Source
VS content of organic waste	$kg_{VS}\ kg^{-1}$	0.27±0.07	lognormal	Calculated based on annex B.5
VS load of blackwater[19]	$kg_{VS}\ p^{-1}\ d^{-1}$	0.047±0.03	lognormal	See annexe B.3
Biogas production	$m^3_{biogas}\ kg_{VS\ in}^{-1}$	0.5±0.1	lognormal	Peters, 2002
Energy content of biogas	$kWh\ m^{-3}_{biogas}$	6.3±0.3	normal	based on Köttner, 2005; Müller et al., 1999; Thomé-Kozmiensky, 1995
Efficiency of the CHP				
$\eta_{thermal}$	-	0.65±0.02	tnormal (0-1)	GLIZIE, 2009
$\eta_{electricity}$	-	0.30±0.02		

Cost parameters related to the Process Anaerobic Digestion

Parameter	Unit	Unit cost (average & stdev)	Lifespan (years)	Distribution	Source
Digester - civil works	€ p^{-1}	138±20	40	normal	Oldenburg and Dlabacs, 2007
Digester - equipment	€ p^{-1}	110±20	12.5	normal	Oldenburg and Dlabacs, 2007
Waste shredding - civil works	€ p^{-1}	2±0.2	40	normal	Oldenburg and Dlabacs, 2007
Waste shredding - equipment	€ p^{-1}	8±0.8	12.5	normal	Oldenburg and Dlabacs, 2007
Maintenance & repair	% of civil works	1.0±0.1		normal	Estimation
Maintenance & repair	% of equipment	3.0±0.3		normal	Estimation
Operation (labour)	€ $p^{-1}\ y^{-1}$	20±5		normal	Balkema, 2003

[19] For blackwater specific VS loads per person instead of concentrations are used in the calculation in order to be able to vary the flush water of the vacuum toilets

A.12 Summarised cost parameters

Summary of cost parameters used for the Hamburg ceMFA

Parameter	Unit	Average	Stdev	Distribution	Source
Rate of interest		**0.03**	0.005	normal	Assumption
Co-digestion organic waste	€ t_{TS}^{-1}	**166**	20	normal	1)
Collection cycle urine, slurry	days	**14**	0	normal	Assumption
Number of residential buildings	-	**235,623**	200	normal	2)
Pipe length in-house per person	m p^{-1}	**2**	0.4	normal	Based on 3)
Number of toilets per household	p^{-1}	**1.3**	0.2	normal	Assumption
Total sewer length (wastewater)	m	**3,764,000**	30,000	normal	4)
Lifespan toilet	y	**20**	0	normal	Assumption
Lifespan pipes	y	**50**	0	normal	Assumption
Lifespan civil works	y	**50**	0	normal	Assumption
Lifespan equipment	y	**12.5**	0	normal	Assumption
Lifespan rainwater infiltration	y	**20**	0	normal	Assumption
Lifespan MBR	y	**20**	0	normal	Assumption
Heat efficiency CHP		**0.65**	0.05	tlognormal (0.40-0.95)	Assumption
Cost electricity	€ kWh^{-1}	**0.15**	0.05	normal	Assumption
Cost thermal energy	€ kWh^{-1}	**0.04**	0.01	normal	Assumption
Benefit CHP	€ kWh^{-1}	**0.08**	0.01	normal	5)
Conventional toilet	€	**275**	25	normal	6) & 7)
Urine diversion toilet	€	**550**	75	normal	6), 7) & 8)
Vacuum toilets	€	**650**	75	normal	6), 7) & 8)
Urine diversion dry toilet	€	**800**	100	normal	9)
Ventilation (dry toilet)	€	**600**	60	normal	7)
Vacuum pipe in-house	€ m^{-1}	**40**	4	normal	6) & 7)
Urine pipe	€ m^{-1}	**20**	2	normal	6) & 7)
Retrofitting existing building	€ hh^{-1}	**1,500**	200	normal	10)
Sewer house connection	€ $build^{-1}$	**900**	50	normal	6)
Vacuum pipe house connection	€ $build^{-1}$	**300**	50	normal	6) & 11)
Urine pipe house connection	€ $build^{-1}$	**300**	50	normal	Assumption
Urine storage tank	€ m^{-3}	**500**	40	normal	6), 12) & 13)
Maintenance urine pipes & composting	-	**0.03**	0.003	normal	6), 7) & 10)
Maintenance pipes, house etc	-	**0.01**	0.0001	normal	Assumption
Operation composting toilet	€ p^{-1} y^{-1}	**15**	3	normal	7) & 14)
Water price	€ m^{-3}	**1.47**	0.02	normal	15)
Water consumption current situation	10^6 m^3 y^{-1}	**68**	10.4	normal	ceMFA model
Fixed water price	€ p^{-1} y^{-1}	**12**	0.1	normal	15)
Bottled water	€ m^{-3}	**127**	13	normal	16)
Conventional sewer	€ m^{-1}	**300**	50	normal	6) & 17)

Parameter	Unit	Average	Stdev	Distribution	Source
Pumping station civil work	€ p^{-1}	**2**	0.4	normal	6)
Pumping station equipment	€ p^{-1}	**2**	0.4	normal	6)
Vacuum sewer	€ m^{-1}	**60**	15	normal	6), 7) & 8)
Vacuum sewer length	m p^{-1}	**1.1**	0.2	normal	Assumption
Vacuum station civil work	€ p^{-1}	**30**	3	normal	6)
Vacuum station equipment	€ p^{-1}	**30**	3	normal	6)
Blackwater storage civil works	€ m^{-3}	**100**	10	normal	6)
Blackwater storage equipment	€ m^{-3}	**25**	2	normal	6)
Sewer maintenance	€ p^{-1} y^{-1}	**40**	4	normal	7)
Vacuum sewer maintenance	€ m^{-1}	**1.3**	0.1	normal	6)
Rainwater infiltraion	€ ha^{-1}	**30,000**	3000	normal	11)
WWTP Q cost	€ m^{-3}	**0.09**	0.009	normal	See section 4.1.5
WWTP COD cost	€ t^{-1}	**150**	15	normal	See section 4.1.5
WWTP N cost	€ t^{-1}	**1160**	116	normal	See section 4.1.5
WWTP P cost	€ t^{-1}	**2360**	236	normal	See section 4.1.5
WWTP re-investment	€ y^{-1}	**19,000,000**	950,000	normal	18)
WWTP investment	€ p^{-1}	**288**	92	normal	19)
Digester (decentralised) civil works	€ p^{-1}	**138**	20	normal	6)
Digester (decentralised) equipment	€ p^{-1}	**110**	20	normal	6)
Waste shredding (decentralised) civil works	€ p^{-1}	**2**	0.02	normal	6)
Waste shredding (decentralised) equipment	€ p^{-1}	**8**	0.08	normal	6)
Digestion (decentralised) O&M civil work	-	**0.01**	0.001	normal	Assumption
Digestion (decentralised) O&M equipment	-	**0.03**	0.003	normal	Assumption
Digester (decentralised) labour	€ p^{-1} y^{-1}	**20**	5	normal	8)
MBR (decentralised)	€ p^{-1}	**400**	50	normal	14), 10), 20) & 21)
MBR (decentralised) O&M	-	**0.025**	0.002	normal	Assumption
UV lamp servicing	€ p^{-1} y^{-1}	**6**	0.5	normal	22)
Transport	€ km^{-1}	**4**	0.5	normal	Based on 1)
Composting	€ t_{TS}^{-1}	**320**	20	normal	1)
Benefit compost	€ t_{TS}^{-1}	**55**	5	normal	1)
Seaborne	€ p^{-1}	**40**	12	normal	23)
Seaborne labour	-	**0.01**	0.001	normal	Assumption
Seaborne O&M	-	**0.03**	0.003	normal	Assumption
NaOH	€ t^{-1}	**300**	60	normal	24) & 25)
H_2SO_4	€ t^{-1}	**200**	40	normal	25) & 27)
MgO	€ t^{-1}	**300**	30	normal	24), 26) & 27)
Benefit NH_4-solution	€ t_N^{-1}	**58**	6	normal	7) 26)
Benefit MAP	€ t_P^{-1}	**1500**	200	normal	7), 28) & 29)
Biocon	€ t_P^{-1}	**3000**	500	normal	Based on 24) & 30)
Steam stripping	€ m^{-3}	**0.722**	0.1	normal	26)

Parameter	Unit	Average	Stdev	Distribution	Source
Steam stripping labour	-	**0.1**	0.01	normal	26)
Steam stripping O&M	-	**0.2**	0.02	normal	26)
Urine MAP	€ t_P^{-1}	**4206**	840	normal	25)
Urine steam stripping	€ t_N^{-1}	**950**	50	normal	26)
Application urine, slurry	€ m^{-3}	**2.5**	20	normal	5) & 31)
Storage urine, slurry	€ m^{-1}	**100**	10	normal	Assumption
Storage urine, slurry, O&M	-	**0.015**	0.001	normal	Assumption
Benefit N sale	€ t_N^{-1}	**1030**	103	normal	25)
Benefit P sale	€ t_P^{-1}	**3130**	313	normal	25)
Benefit K sale	€ t_K^{-1}	**600**	60	normal	25)
Benefit S sale	€ t_S^{-1}	**360**	36	normal	25)
Benefit H_3PO_4	€ t_{H3PO4}^{-1}	**3110**	311	normal	32)

Sources: 1) Arlt, 2003, 2) Statistisches Amt für Hamburg und Schleswig-Holstein, 2008a, 3) Buchert et al., 2004, 4) Statistikamt Nord, 2009, 5) BMU, 2009, 6) Oldenburg and Dlabacs, 2007, 7) Herbst, 2008, 8) Balkema, 2003, 9) Werner et al., 2006, 10) Starkl et al., 2005, 11) Halbach, 2003, 12) Görges, 2008, 13) Drücker, 2004, 14) Kaufmann et al., 2007, 15) Hamburg Wasser, 2009, 16) Destatis, 2006, 17) Reicherter, 2001, 18) BSU, 2009, 19) Reicherter, 2001, 20) fbr, 2005, 21) Kionka, 2008, 22) Boller, 2006, 23) Seaborne EPM AG, 2009, 24) Fehr, 2007, 25) Esemen and Dockhorn, 2009, 26) Dockhorn, 2007, 27) Ek et al., 2006, 28) Montag, 2008, 29) Wilsenach, 2006, 30) Stark, 2002, 31) Effenberger et al., 2006, 32) Linker, 2008

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Annex B: Characteristic values of source-separated flows

Annex B illustrates the results of an extensive literature review on 157 references regarding wastewater flows and on 72 references regarding organic waste flows carried out to generate standard characteristic values of source-separated wastewater flows, such as urine, faeces and greywater. A list of all references used for the characteristics of urine, faeces, blackwater and greywater is included at the end of Annex B. The references used for the characteristics of organic waste are based on a database of the consultancy OtterWasser GmbH and are not included here. Only sources from European studies are used for the generation of characteristic values, since nutritional and water use habits vary too much across the continents. In addition, data availability from other continents is not sufficient for specific values. For a more detailed discussion of regional variances, please refer to Meinzinger and Oldenburg (2009). The data quality is assessed using the number of original datasets as indicator. A good data quality is assigned when the number of original values is more than 15, whereas less than five data sources are considered as poor data quality. Please note that the values differ from values given in Meinzinger and Oldenburg (2009), since averages and not median values are shown here. Data on loads and concentrations do not always fully correlate, since they are often derived from different sources.

B.1 Loads and concentrations in urine

Parameter		Unit	Average	Stdev	Data quality	Unit	Average	Stdev	Data quality
Volume		$l\ p^{-1}\ d^{-1}$	**1.27**	0.32	++				
TS[20]		$g\ p^{-1}\ d^{-1}$	**45.4**	19.7	++	$g\ l^{-1}$	**13.9**	13.0	-
VS		$g\ p^{-1}\ d^{-1}$	**34.8**	15.1	+	$g\ l^{-1}$	**-**	-	-
Org. matter	BOD_5	$g\ p^{-1}\ d^{-1}$	**4.9**	1.2	+	$g\ l^{-1}$	**3.9**	-	-
	COD	$g\ p^{-1}\ d^{-1}$	**9.8**	3.3	+	$g\ l^{-1}$	**6.9**	2.6	+
	TOC	$g\ p^{-1}\ d^{-1}$	**5.3**	3.1	+	$g\ l^{-1}$	**3.7**	0.6	-
Nutrients	N	$g\ p^{-1}\ d^{-1}$	**9.5**	2.6	++	$g\ l^{-1}$	**6.3**	2.8	++
	P	$g\ p^{-1}\ d^{-1}$	**1.0**	0.4	++	$g\ l^{-1}$	**0.7**	0.5	++
	K	$g\ p^{-1}\ d^{-1}$	**2.4**	0.6	++	$g\ l^{-1}$	**1.9**	0.6	++
	S	$g\ p^{-1}\ d^{-1}$	**0.8**	0.3	+	$g\ l^{-1}$	**0.9**	0.7	-

Data quality: - poor (or data not available) + fair ++ good

[20] Some of the analysed references do not particularly state whether TS (total solids) or TSS (total suspended solids) were analysed. Yet, it is assumed for all flows that parameters such as dry matter, dry residues, dry solids etc. refer to TS. The same applies to VS (volatile solids) compared to VSS (volatile suspended solids).

B.2 Loads and concentrations in faeces

	Parameter	Unit	Average	Stdev	Data quality	Unit	Average	Stdev	Data quality
	Volume	$l\ p^{-1}\ d^{-1}$	**0.15**	0.04	++				
	TS	$g\ p^{-1}\ d^{-1}$	**40.4**	10.8	++	$g\ l^{-1}$	**-**	-	-
	VS	$g\ p^{-1}\ d^{-1}$	**34.4**	13.6	+	$g\ l^{-1}$	**-**	-	-
Org. matter	BOD_5	$g\ p^{-1}\ d^{-1}$	**19.5**	0.6	-	$g\ l^{-1}$	**-**	-	-
Org. matter	COD	$g\ p^{-1}\ d^{-1}$	**50.0**	11.7	+	$g\ l^{-1}$	**-**	-	-
Org. matter	TOC	$g\ p^{-1}\ d^{-1}$	**26.2**	10.7	+	$g\ l^{-1}$	**-**	-	-
Nutrients	N	$g\ p^{-1}\ d^{-1}$	**1.7**	0.7	++	$g\ l^{-1}$	**9.9**	7.0	-
Nutrients	P	$g\ p^{-1}\ d^{-1}$	**0.6**	0.4	++	$g\ l^{-1}$	**3.4**	1.1	-
Nutrients	K	$g\ p^{-1}\ d^{-1}$	**0.7**	0.3	++	$g\ l^{-1}$	**3.3**	0.1	-
Nutrients	S	$g\ p^{-1}\ d^{-1}$	**0.2**	-	-	$g\ l^{-1}$	**-**	-	-

Data quality: - poor (or data not available) + fair ++ good

B.3 Loads in blackwater

Blackwater is defined as the mixture of urine, faeces, toilet paper and flush water. No concentrations are given here, since the database is not sufficient to reflect the variances in flush water volume. The data is not necessarily the same as the sum of faeces and urine, since it is derived from different sources. However, it is generally in the same range.

	Parameter	Unit	Average	Stdev	Data quality	Unit	Average	Stdev	Data quality
	Volume	$l\ p^{-1}\ d^{-1}$	**49.0**	39.6	-				
	TS	$g\ p^{-1}\ d^{-1}$	**63.2**	41.3	+	$g\ l^{-1}$	**-**	-	-
	VS	$g\ p^{-1}\ d^{-1}$	**47.3**	30.2	-	$g\ l^{-1}$	**-**	-	-
Org. matter	BOD	$g\ p^{-1}\ d^{-1}$	**32.4**	10.8	-	$g\ l^{-1}$	**-**	-	-
Org. matter	COD	$g\ p^{-1}\ d^{-1}$	**51.5**	31.1	+	$g\ l^{-1}$	**-**	-	-
Org. matter	TOC	$g\ p^{-1}\ d^{-1}$	**27.9**	5.2	-				
Nutrients	N	$g\ p^{-1}\ d^{-1}$	**10.7**	2.7	+	$g\ l^{-1}$	**-**	-	-
Nutrients	P	$g\ p^{-1}\ d^{-1}$	**1.5**	0.6	+	$g\ l^{-1}$	**-**	-	-
Nutrients	K	$g\ p^{-1}\ d^{-1}$	**2.7**	1.3	+	$g\ l^{-1}$	**-**	-	-
Nutrients	S	$g\ p^{-1}\ d^{-1}$	**-**	-	-	$g\ l^{-1}$	**-**	-	-

Data quality: - poor (or data not available) + fair ++ good

B.4 Loads and concentrations in greywater

Greywater is here defined as total wastewater from kitchens and bathrooms without any toilet wastewater.

	Parameter	Unit	Average	Stdev	Data quality	Unit	Average	Stdev	Data quality
	Volume	$l\ p^{-1}\ d^{-1}$	**105**	26	++				
	TS	$g\ p^{-1}\ d^{-1}$	**59.5**	33.3	++	$mg\ l^{-1}$	**159**	98	-
Org. matter	BOD_5	$g\ p^{-1}\ d^{-1}$	**17.9**	9.2	+	$mg\ l^{-1}$	**305**	75	+
Org. matter	COD	$g\ p^{-1}\ d^{-1}$	**47.7**	21.3	++	$mg\ l^{-1}$	**551**	132	+
Org. matter	TOC	$g\ p^{-1}\ d^{-1}$	**13.3**	2.0	-	$mg\ l^{-1}$	**213**	-	-
Nutrients	N	$g\ p^{-1}\ d^{-1}$	**1.0**	0.4	++	$mg\ l^{-1}$	**16.3**	5.6	+
Nutrients	P	$g\ p^{-1}\ d^{-1}$	**0.5**	0.3	++	$mg\ l^{-1}$	**5.4**	2.5	+
Nutrients	K	$g\ p^{-1}\ d^{-1}$	**1.3**	1.1	++	$mg\ l^{-1}$	**8.8**	-	-
Nutrients	S	$g\ p^{-1}\ d^{-1}$	**3.5**	2.7	+	$mg\ l^{-1}$	**72**	-	-

Data quality: - poor (or data not available) + fair ++ good

B.5 Loads and concentrations in organic waste (generic)

The following table provides information on general organic waste from households, i.e. no differentiation between kitchen waste or garden waste is made.

	Parameter	Unit	Average	Stdev	Data quality	Unit	Average	Stdev	Data quality
	mass	$g\ p^{-1}\ d^{-1}$	**271.8**	119.4	++				
	density	$t\ m^{-3}$	**0.61**	0.3	-				
	TS	$g\ p^{-1}\ d^{-1}$	**105.0**	56.7	+	%weight	**39.0**	16.3	++
	VS	$g\ p^{-1}\ d^{-1}$	**64.9**	21.3	+	%TS	**69.2**	17.9	++
	TOC	$g\ p^{-1}\ d^{-1}$	**29.2**	1.8	-	%TS	**44.0**	6.3	+
Nutrients	N	$g\ p^{-1}\ d^{-1}$	**1.7**	0.1	-	$mg\ gTS^{-1}$	**17.3**	5.8	++
Nutrients	P	$g\ p^{-1}\ d^{-1}$	**0.4**	0.1	-	$mg\ gTS^{-1}$	**3.0**	1.3	++
Nutrients	K	$g\ p^{-1}\ d^{-1}$	**0.9**	0.1	-	$mg\ gTS^{-1}$	**7.7**	2.2	+
Nutrients	S	$g\ p^{-1}\ d^{-1}$	**-**	-	-	$mg\ gTS^{-1}$	**2.3**	0.2	-

Data quality: - poor (or data not available) + fair ++ good

B.6 Loads and concentrations in kitchen organic waste

The following table provides information on domestic organic waste from kitchens.

	Parameter	Unit	Average	Stdev	Data quality	Unit	Average	Stdev	Data quality
	mass	g p^{-1} d^{-1}	**185.5**	75.4	++				
	TS	g p^{-1} d^{-1}	**44.0**	7.7	+	%weight	**23.3**	6.0	+
	VS	g p^{-1} d^{-1}	**37.7**	7.9	+	%TS	**72.9**	24.4	+
	TOC	g p^{-1} d^{-1}	**15.3**	4.5	-	%TS	**-**	-	-
Nutrients	N	g p^{-1} d^{-1}	**1.0**	0.3	+	mg gTS^{-1}	**19.4**	5.2	+
Nutrients	P	g p^{-1} d^{-1}	**0.2**	0.1	+	mg gTS^{-1}	**4.4**	1.1	+
Nutrients	K	g p^{-1} d^{-1}	**0.4**	0.2	-	mg gTS^{-1}	**10.4**	1.4	+
Nutrients	S	g p^{-1} d^{-1}	**0.1**	-	-	mg gTS^{-1}	**-**	-	-

Data quality: - poor (or data not available) + fair ++ good

References Annex B

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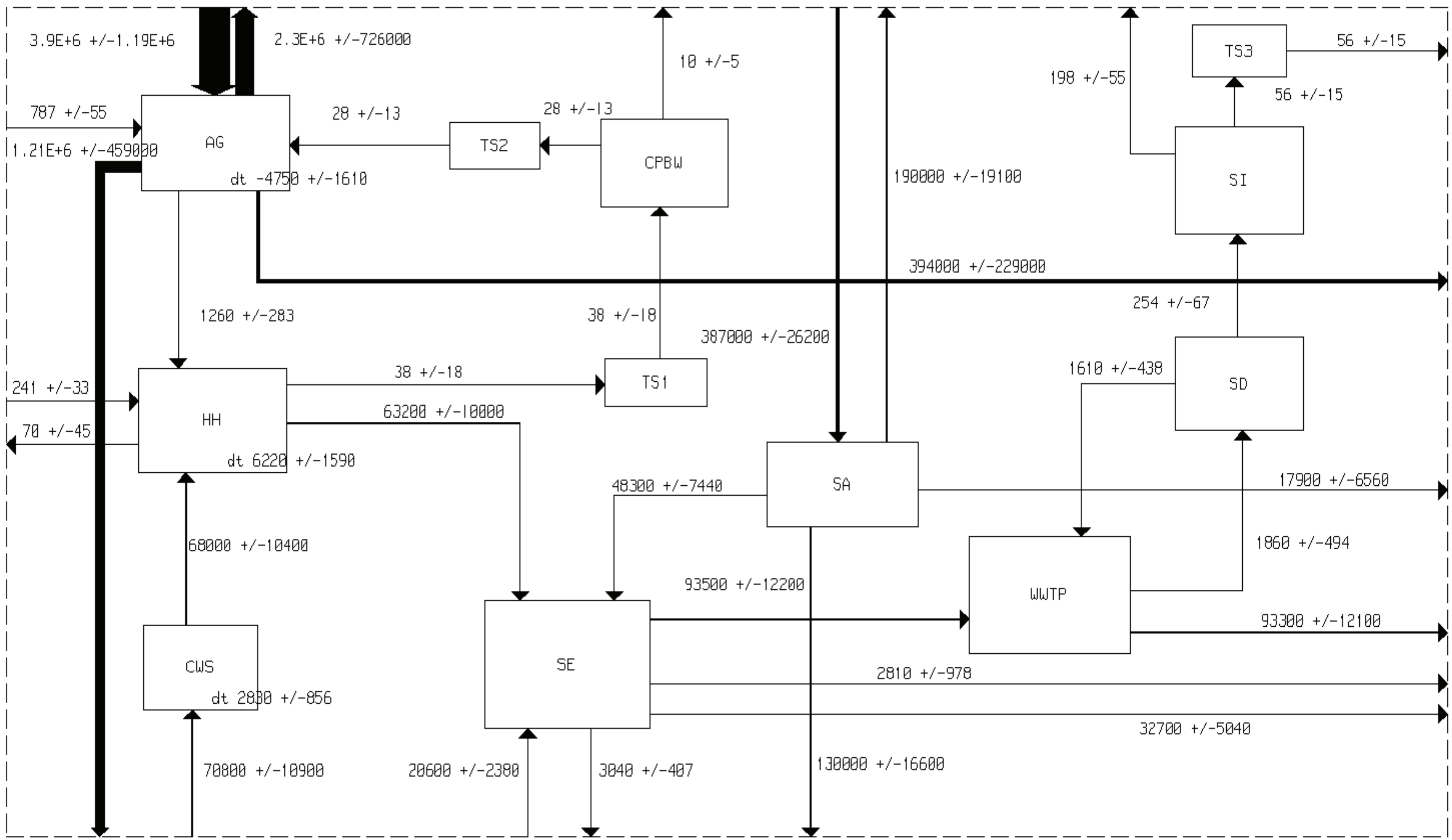
Annex C: Flow schemes of system 1 CurS (Hamburg)

The modelling in SIMBOX results in the representation of the flows in flow schemes similar to Sankey diagrams, where the witdth of the arrows is proportional to the flow quantity. This allows grasping important information at a glance. In order to limit the length of this thesis, the following pages exemplify flow schemes only for one system, i.e. System 1 CurS, and for certain flows, i.e. mass flows, nitrogen flows and phosphorus flows. The flow schemes of the other substances and the other systems of the case study Hamburg can be requested at ceMFA@ymail.com.

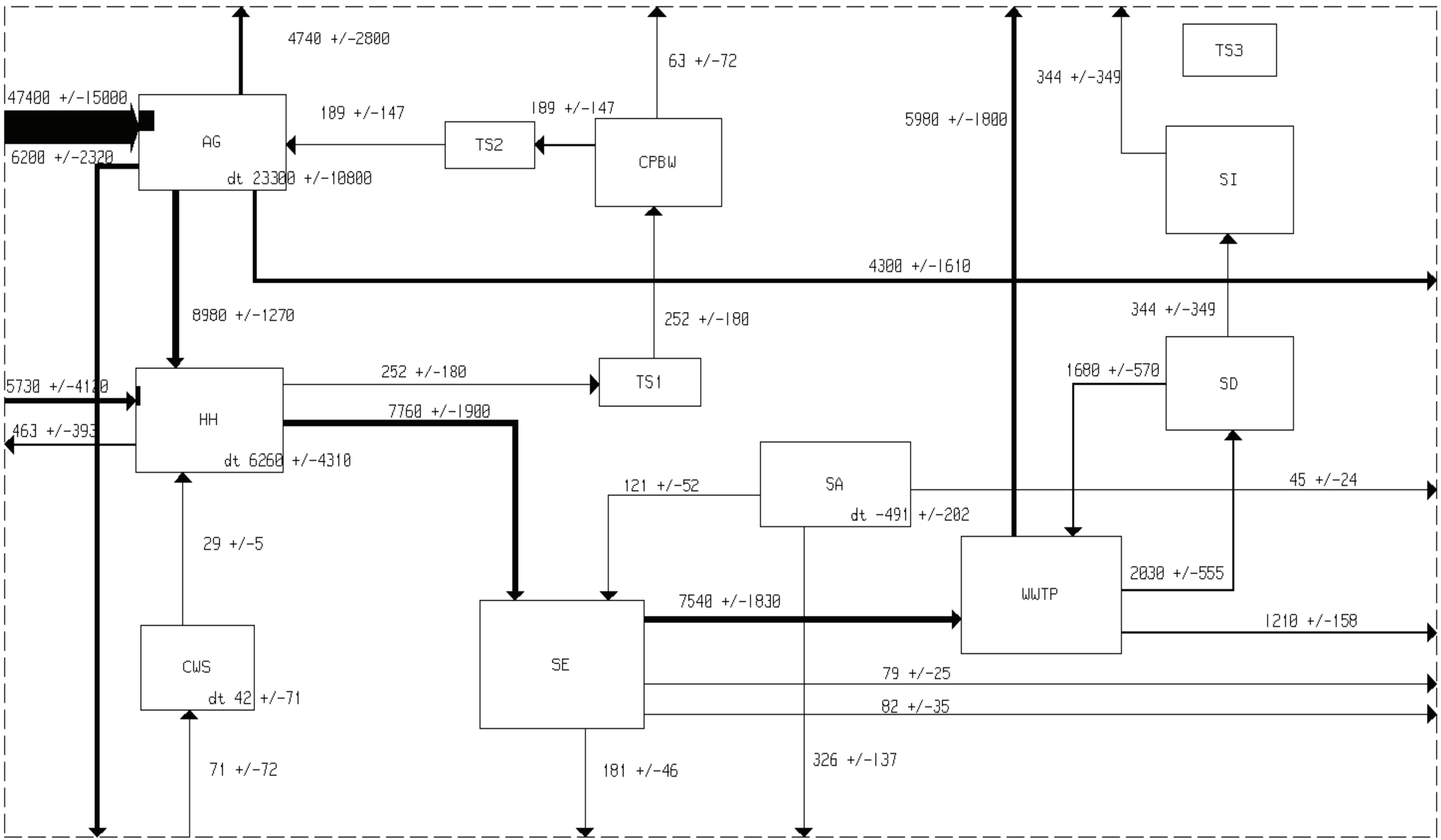
The general system layout including the names of all processes and flows are depicted in detail in Section 3.5.2. In Annex C, following abbreviations are used in the flow scheme diagrammes for the processes:

AG	Agriculture
CPBW	Composting of biowaste
CWS	Central water supply
HH	Households
SA	Surface areas
SD	Sludge Digestion
SE	Sewerage
SI	Sludge incineration
TS	Transport
WWTP	Wastewater treatment plant

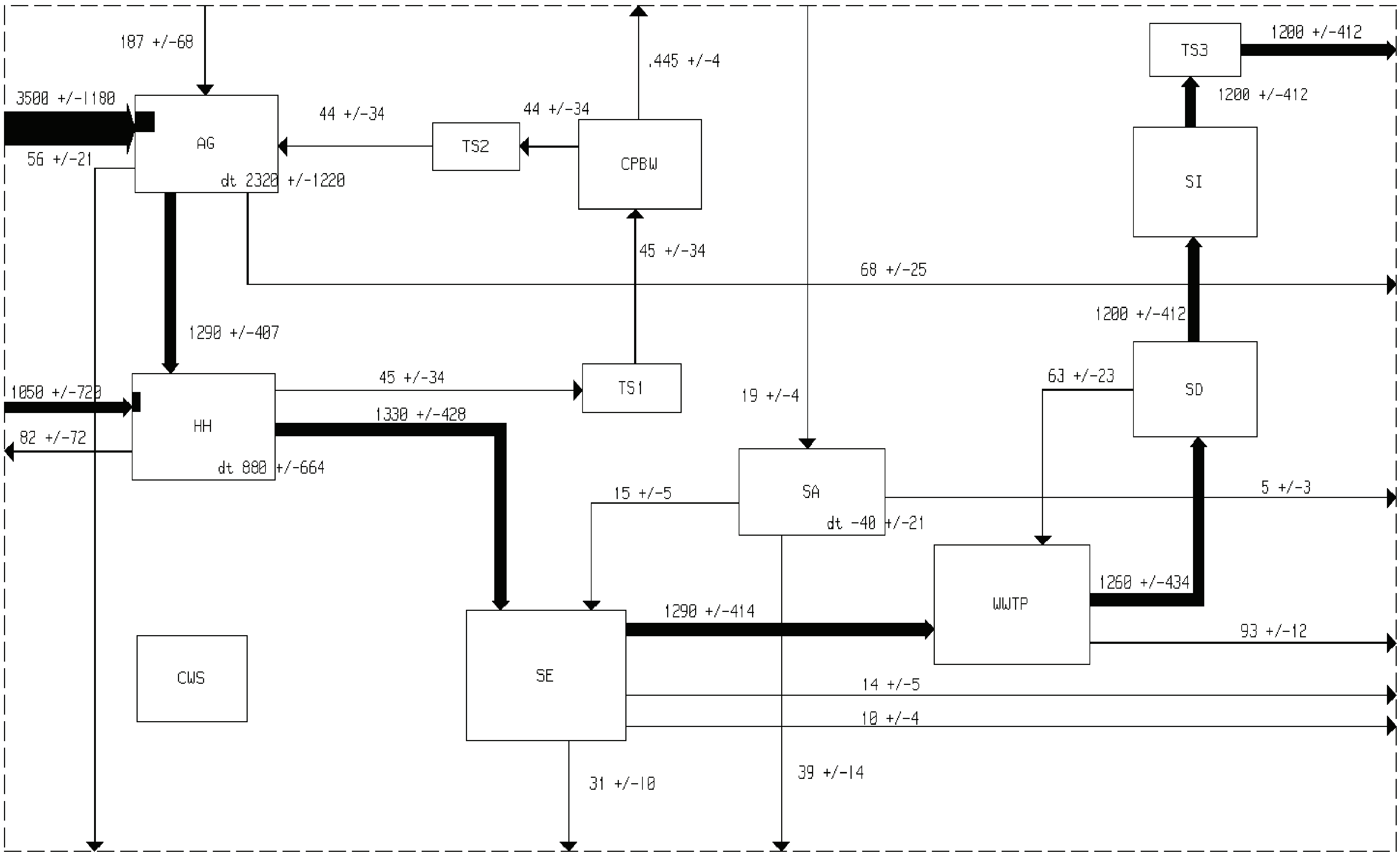
1 CurS - Mass flows [1000 t y^{-1}]



1 CurS - Nitrogen flows [t y-1]



1 CurS - Phosphorus flows [t y^{-1}]

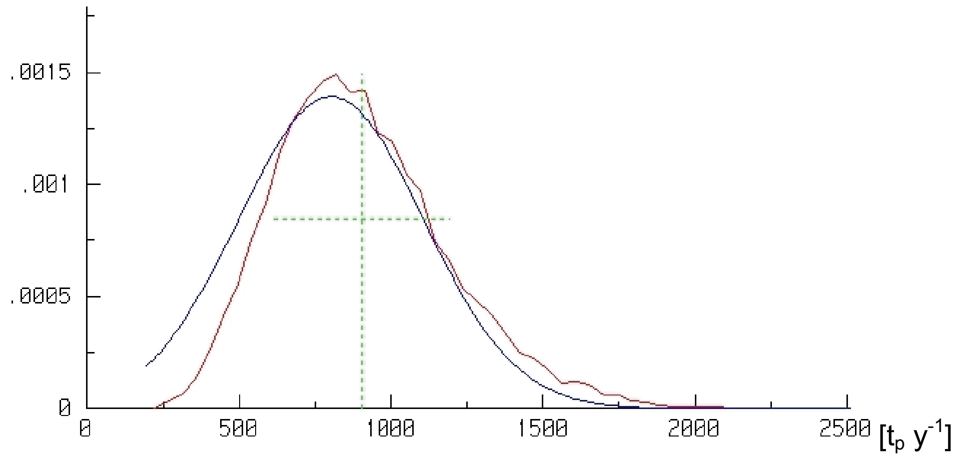


Annex D: Monte Carlo simulations – recovered P in Hamburg systems

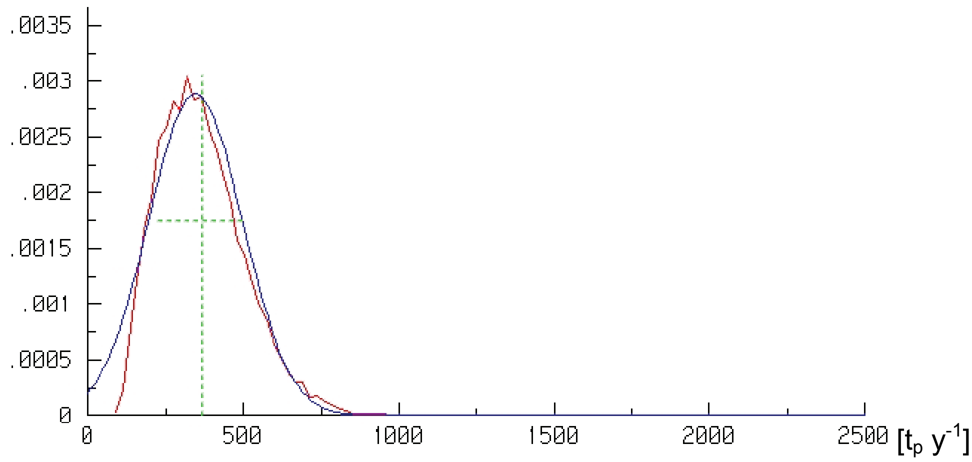
SIMBOX allows the simulation of the ceMFA results using Monte Carlo methods. As described in Section 3.3, for every equation system uncertainties are calculated based on Gauss' law of error propagation, but in addition also Monte Carlo simulations are applied. For the Monte Carlo calculations random samples with a sample size of 20,000 are generated. The simulations are carried out using the uncertainty distributions assigned to the parameters as shown in Annexes A and F.

Overall, the observed differences between Monte Carlo results and the results using Gaussian distribution are rather small. In order to limit the length of this thesis, only a very narrow selection of Monte Carlo results can be shown here. As example, the probability distribution of recovered phosphorus in the Hamburg systems as it results from Monte Carlo simulations and from Gaussian distribution is illustrated in the following figures. Further results from the Monte Carlo simulations can be requested from ceMFA@ymail.com.

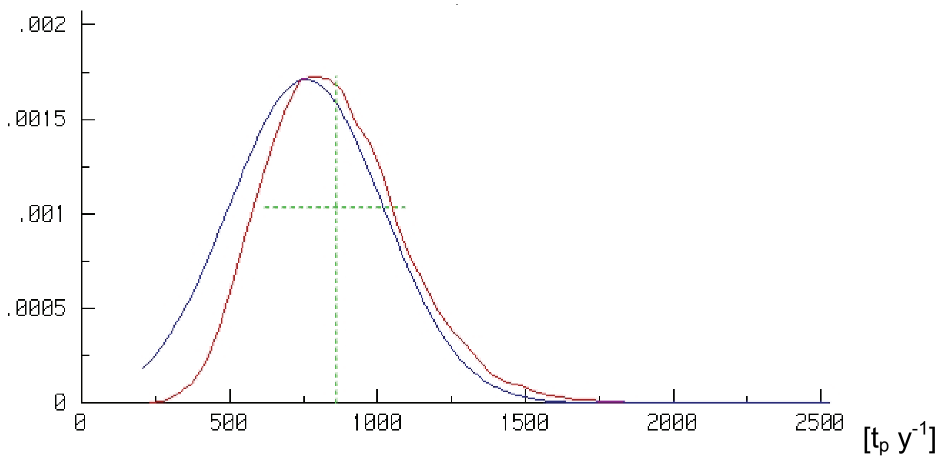
2 NurS



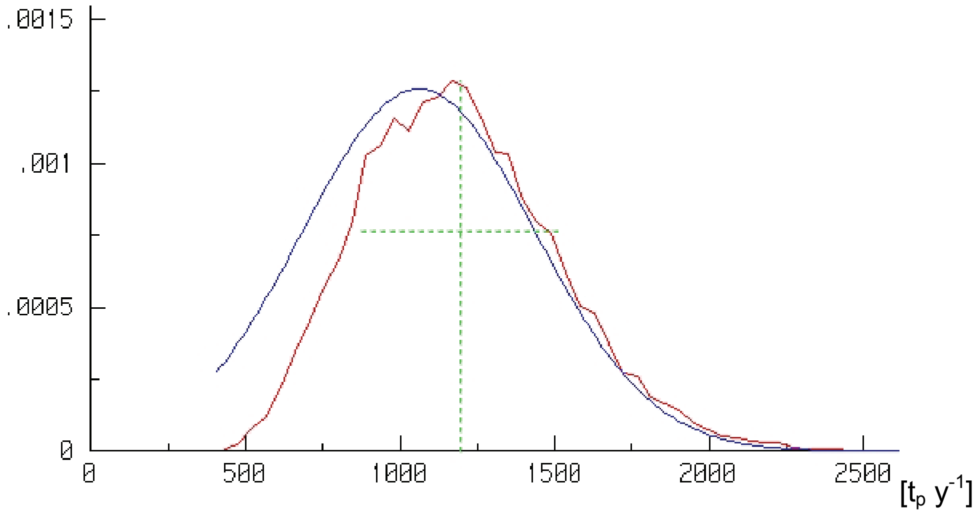
3 NuRU



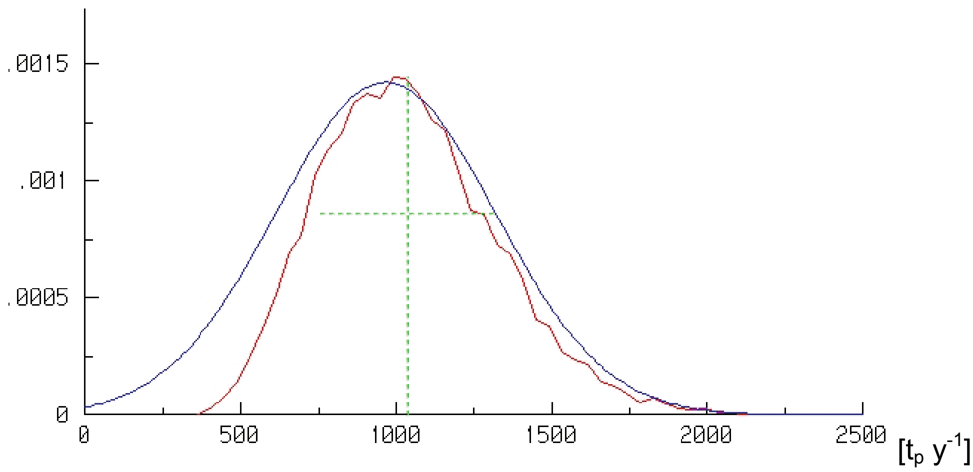
4 CoDig



5 BlaD



6 CompU



Annex E: Sensitivities of the Hamburg systems

Annex E shows the key parameters (up to ten are included) for each of the criteria nutrient recovery, energy demand per person and costs per person (case of existing infrastructure). The relative sensitivity of every parameter is shown in percent, i.e. a parameter change of 10% results in a variable change of the shown percentage. A negative value indicates that if the value of the parameter is reduced, then the value of the variable increases. Only relative sensitivities greater than 1% or smaller than -1% are included.

2 NurS	Rate	3 NuRU	Rate	4 HWC	Rate	5 BlaD	Rate	6 CompU	Rate
Recovered Nitrogen (values indicate the percentage of variable change at 10% parameter change)									
Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0
N incorporated in biomass (WWTP)	6.7	Rate of urine collection	10.0	N transfer sludge liquor	10.0	N in urine	8.2	N in urine	10.0
Ratio COD/TOC in wastewater	6.4	N in urine	10.0	Degree of N recovery (stripping)	10.0	N in faeces	1.5	Rate of urine collection	10.0
TOC in faeces	4.1	Degree of N recovery (stripping)	9.6	N in urine	7.5				
N removed in primary sludge	3.7			N in faeces	1.4				
N in urine	2.8								
TOC in greywater	2.1								
TOC removed in primary sludge	-3.0								
Recovered Phosphorus (values indicate the percentage of variable change at 10% parameter change)									
Inhabitants of Hamburg	10.2	P in urine	10.0	Inhabitants of Hamburg	10.2	Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0
Degree of P recovery (Seaborne)	10.0	Rate of urine collection	10.0	Degree of P recovery (BioCon)	10.0	P in urine	6.0	Rate of urine collection	10.0
P in urine	5.1	Inhabitants of Hamburg	10.0	P in urine	4.9	P in faeces	3.6	P in urine	10.0
P in faeces	3.1	Degree of P recovery (MAP)	10.0	P in faeces	2.9			P loss during storage	-2.5
P in greywater	2.6	P loss during storage	-2.5	P in greywater	2.5				
Recovered Potassium (values indicate the percentage of variable change at 10% parameter change)									
K in organic waste	10	not applicable		not applicable		Inhabitants of Hamburg	10.0		
Inhabitants of Hamburg	10					K in urine	7.3	Rate of urine collection	10.0
Organic waste per person	10					K in faeces	2.1	Inhabitants of Hamburg	10.0
TS in organic waste	10							K in urine	10.0
Rate of organic waste collection	10								
Recovered Sulphur (values indicate the percentage of variable change at 10% parameter change)									
Sulphur for stripping	10.0	Sulphur for stripping	10.0	Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0	Inhabitants of Hamburg	10.0
Inhabitants of Hamburg	10.0			N transfer sludge liquor	10.0	S in urine	7.6	S in urine	10.0
N incorporated in biomass (WWTP)	8.3			Degree of N recovery (stripping)	10.0	S in faeces	1.8	Rate of urine collection	10.0

2 NurS	Rate	3 NuRU	Rate	4 HWC	Rate	5 BlaD	Rate	6 CompU	Rate
Ratio COD/TOC in wastewater	7.9			N in urine	7.5				
TOC in faeces	5.0			N in faeces	1.4				
N removed in primary sludge	4.5			Sulphur for stripping	10.0				
P in urine	-1.2								
Degree of N recovery (MAP)	-2.3								
Degree of P recovery (Seaborne)	-2.3								
TOC removed in primary sludge	-3.7								
Cost per person (existing infrastructure setup) (values indicate the percentage of variable change at 10% parameter change)									
Water price (public supply)	2.8	Number of households	2.5	Number of households	2.1	Water price (bottled)	5.5	Number of households	3.7
Water price (bottled)	2.7	Consumption bottled water	2.1	Consumption bottled water	1.9	Consumption bottled water (decentralised)	5.5	Toilets per household	2.3
Consumption bottled water	2.7	Water price (bottled)	2.1	Water price (bottled)	1.9	Number of households	1.1	Water price (bottled)	2.1
Fixed costs public water supply	2.2	Water price (public supply)	2.1	Water price (public supply)	1.9	Inhabitants of Hamburg	-1.6	Consumption bottled water	2.1
O&M sewer system	1.9	Fixed costs public water supply	1.8	Fixed costs public water supply	1.6			Sewered area	1.9
Inhabitants of Hamburg	-3.0	O&M sewer system	1.5	O&M sewer system	1.4			Rate of interest	1.9
		Toilets per household	1.4	Cost vacuum toilet	1.1			Cost MBR	1.7
		Cost urine-diversion toilet	1.4	Toilets per household	1.1			Cost UDD toilet	1.6
		Cost retrofitting	1.1	Cost retrofitting	1.0			Area with separate sewer	-1.3
		Inhabitants of Hamburg	-4.7	Inhabitants of Hamburg	-4.8			Inhabitants of Hamburg	-4.5
Energy per person (values indicate the percentage of variable change at 10% parameter change)									
Area per 1kg food	6.7	Area per 1kg food	7.6	Area per 1kg food	6.1	Area per 1kg food	4.5	Area per 1kg food	6.5
Food per person	6.7	Food per person	7.6	Food per person	6.1	Food per person	4.5	Food per person	6.5
N per hectare	6.4	N per hectare	7.1	N per hectare	5.8	N per hectare	4.2	N per hectare	6.1
Energy N production	6.1	Energy N production	6.6	Energy N production	5.1	Energy N production	3.6	Energy N production	5.3
Conversion primary energy	3.6	Conversion primary energy	3.1	Conversion primary energy	4.4	Conversion primary energy	3.1	Conversion primary energy	4.2
Energy sludg drying	1.2	Energy sludge drying	1.6	Flush vacuum toilet	2.1	Flush vacuum toilet	2.5	Energy greywater treatment (incl. disinfection)	2.5

2 NurS	Rate	3 NuRU	Rate	4 HWC	Rate	5 BlaD	Rate	6 CompU	Rate
Greywater per person	1.1	Greywater per person	1.3	Energy vacuum sewer	1.7	Energy bottled water	2.1	Greywater per person	2.2
Energy water supply (public)	1.0	Biogas per kg VS	-1.2	Energy sludge drying	1.6	Consumption bottled water (decentralised)	2.1	Energy forced ventilation	1.6
Lower heating value dried sludge	-1.1	Lower heating value dried sludge	-1.6	Lower heating value dried sludge	-1.9	Temperature pasteurisation	2.0	Number of households	1.6
Efficiency energy production sludge treatment	-2.0	Efficiency energy production sludge treatment	-2.8	Efficiency energy production sludge treatment	-3.1	Heat capacity water	1.4	Inhabitants of Hamburg	-1.6

Annex F: Parameters of the Arba Minch ceMFA model

Annex F summarises the data that was used as input parameters for the development of the ceMFA model for the town of Arba Minch. The data is listed according to the processes as they are described in section 5.2.

For the Monte Carlo simulations, the distribution of the parameter values can be selected to show different distributions, namely normal, tnormal (i.e. truncated normal), lognormal, tlognormal (i.e. truncated lognormal), and uniform. If a truncated distribution is selected, the minium and maximum values are included.

F.1 Process Agriculture

Parameters used to calculate mass, nutrient, energy and cost flows of the Process Agriculture

Parameter	Unit	Average & Stdev	Distribution	Source
Rainfall	mm y^{-1}	888±100	normal	NMASZ, 2006
Evaporation	mm y^{-1}	800±100	normal	Deichmann and Eklundh, 1991
Plant water demand	mm y^{-1}	415±40	normal	Awulachew, 2001
N fertiliser demand	kg_N ha^{-1} y^{-1}	39±6	normal	Plückers, 2009
P fertiliser demand	kg_P ha^{-1} y^{-1}	20.1±1	normal	Plückers, 2009
Primary energy demand N production	kWh kg_N^{-1}	13.64±2.73	normal	Patyk and Reinhardt, 1997
Primary energy demand P production	kWh kg_P^{-1}	2.14±0.43	normal	Patyk and Reinhardt, 1997

F.2 Process Households

Parameters used to calculate mass, nutrient, energy and cost flows of the Process Households

Parameter	Unit	Average & Stdev	Distribution	Source
Food consumption	kg p^{-1} y^{-1}	290±30	normal	FAO, 2009
Faeces	kg p^{-1} d^{-1}	0.30±0.03	tnormal (0.1-0.4)	Jönsson and Vinneras, 2004
Urine	l p^{-1} d^{-1}	1.1±0.1	tnormal (0.5-2.0)	Assumption
Urine collection rate (i.e. efficiency of UDDT)	-	0.7±0.1	tnormal (0.1-1.0)	Assumption
Organic waste	kg p^{-1} d^{-1}	0.12±0.01	lognormal	Kuma, 2004
N in food	g_N p^{-1} y^{-1}	2411±240	normal	FAO, 2009; Souci et al., 1995
N in organic waste	g_N kg_{TS}^{-1}	17.3±5.8	normal	See Annex B

Parameter	Unit	Average & Stdev	Distribution	Source
N in excreta	$g_N\ p^{-1}\ d^{-1}$	6.94±1.0	normal	Calculated based on FAO, 2009; Jönsson and Vinneras, 2004
N in greywater	$g_N\ p^{-1}\ d^{-1}$	0.8±0.3	tnormal (0-3.0)	Ridderstolpe, 2004
P in food	$g_P\ p^{-1}\ y^{-1}$	432±40	normal	FAO, 2009; Souci et al., 1995
P in organic waste	$g_P\ kg_{TS}^{-1}$	3.0±1.3	tnormal (0.9-6.1)	See Annex B
P in excreta	$g_P\ p^{-1}\ d^{-1}$	1.11±0.10	tnormal (0-3.0)	Calculated based on FAO, 2009; Jönsson and Vinneras, 2004
P in greywater	$g_P\ p^{-1}\ d^{-1}$	0.50±0.05	tnormal (0.2-1.0)	Ridderstolpe, 2004
VS in blackwater	$g_{VS}\ p^{-1}\ d^{-1}$	47.3±30.2	normal	See Annex B
TS in organic waste	-	0.39±0.04	tnormal (0.1-0.8)	See Annex B
VS in organic waste	-	0.26±0.02	tnormal (0-1.0)	See Annex B
C in urine	$g_{TOC}\ p^{-1}\ d^{-1}$	5.3±3.1	normal	See Annex B
C in faeces	$g_{TOC}\ p^{-1}\ d^{-1}$	26.2±10.7	normal	See Annex B
C in greywater	$g_{TOC}\ p^{-1}\ d^{-1}$	13.2±2.0	normal	See Annex B
C in food	$g_{TOC}\ p^{-1}\ y^{-1}$	17.52±1.8	normal	Based on FAO, 2009; Wu Leung et al., 1968
C in organic waste	$kg_{TOC}\ kg_{TS}^{-1}$	0.44±0.06	lognormal	See Annex B
Fuelwood consumption	$kg\ p^{-1}\ y^{-1}$	120±24	normal	Abebaw, 2007
Charcoal consumption	$kg\ p^{-1}\ y^{-1}$	120±24	normal	Abebaw, 2007
Lower heating value fuelwood	$kWh\ kg^{-1}$	4±0.2	normal	Gloor, 2009
Lower heating value charcoal	$kWh\ kg^{-1}$	8±0.2	normal	Gloor, 2009
Efficiency of wood stoves	-	0.2±0.05	tnormal (0-1.0)	Addison, 2010
Efficiency of biogas stoves	-	0.6±0.05	tnormal (0-1.0)	Seyoum, 1988
Cost firewood	$ETB\ kg^{-1}$	0.3±0.03	normal	Based on Abebaw, 2007; Mekonnen and Köhlin, 2008
Cost charcoal	$ETB\ kg^{-1}$	0.8±0.08	normal	Based on Tamru, Z. (2010), ROSA, pers.communication
Cost rainwater storage tank ($2m^3$)	ETB	3400±300	normal	Based on market prices
Lifespan rainwater storage tank	y	15±1	normal	Assumption
O&M rainwater harvesting as % of investment	-	0.02±0.01	tnormal (0-1.0)	Assumption
Cost greywater tower	ETB	400±40	normal	Seyoum, 2008
Lifespan greywater tower	y	2±1	tnormal (0-10)	Seyoum, 2008

F.3 Process Water Supply

Parameters used to calculate mass, nutrient, energy and cost flows of the Process Water Supply

Parameter	Unit	Average & Stdev	Distribution	Source
Pump head	m	350±50	normal	Assumption (based on Gelaye, B., pers. comm., Water Supply Service)
Pump efficiency	-	0.6±0.1	tnormal (0-1.0)	Assumption
Cost of water	ETB m^{-3}	1.5±0.2	normal	AMU and ARB, 2007

F.4 Process University

Parameters used to calculate mass, nutrient and energy flows of the Process University (AMU)

Parameter	Unit	Average & Stdev	Distribution	Source
Residents at AMU	p	7,429±700	normal	Kröger, 2007
Water supply to AMU	l p^{-1} d^{-1}	46±5	normal	Based on Dagalo, 2005; Kröger, 2007
Water loss at AMU	-	0.22±0.02	tnormal (0-1.0)	Kröger, 2007
Pump head water supply	m	30±5	normal	AMU and ARB, 2007
Area of pond system	m^2	3,000±300	normal	Kröger, 2007
Evaporation	mm y^{-1}	800±100	normal	Deichmann and Eklundh, 1991
Rainfall	mm y^{-1}	887±90	normal	NMASZ, 2006
Pond water used for irrigation	-	0.5±0.2	uniform (0.3-0.7)	Assumption
N elimination in pond	-	0.5±0.1	tnormal (0-1.0)	Assumption
P elimination in pond	-	0.55±0.07	tnormal (0-1.0)	Gutzeit and Neis, 2007
C elimination in pond	-	0.9±0.5	tnormal (0-1.0)	Assumption
Lifespan	y	40±4	normal	Assumption
O&M costs as ratio of construction costs	-	0.025±0.003	tlognormal (0-1.0)	Loetscher, 2002

F.5 Process On-Site Sanitation Facilities

Parameters used to calculate mass, nutrient, energy and cost flows of the Process On-site Sanitation Facilities

Parameter	Unit	Average & Stdev	Distribution	Source
Faecal sludge production (anaerobic treatment)	kg p^{-1} d^{-1}	0.25±0.05	lognormal	Assumption, based on Montangero et al., 2005
N volatilisation in UDDT	-	0.08±0.01	tlognormal (0-0.8)	Montangero and Belevi, 2007
N transfer into septage/slurry	-	0.08±0.01	tnormal (0-1.0)	Belevi, 2002
N transfer into pit latrine sludge	-	0.17±0.02	tnormal (0-1.0)	Montangero and Belevi, 2007
P transfer into septage/slurry	-	0.17±0.02	tnormal (0-1.0)	Montangero and Belevi, 2007
P transfer into pit latrine sludge	-	0.28±0.03	tnormal (0-1.0)	Montangero and Belevi, 2007
C transfer into septage/slurry	-	0.43±0.05	tnormal (0-1.0)	Wendland, 2008
CH_4 in biogas	-	0.65±0.05	tnormal (0-1.0)	Smil, 1983
Biogas production	m^3 kg_{VS}^{-1}	0.5±0.0.05	tnormal (0-1.0)	Müller et al., 1999
Heating value biogas	MJ m^3	22.5±1.0	normal	Thomé-Kozmiensky, 1995
Water loss in soil filter	-	0.2±0.1	tnormal (0-1.0)	Assumption
N elimination in soil filter	-	0.4±0.15	tnormal (0-1.0)	Assumption
P elimination in soil filter	-	0.6±0.15	tnormal (0-1.0)	Assumption
C elimination in soil filter	-	0.7±0.15	tnormal (0-1.0)	Assumption
Investment pit latrine	ETB	1,500±500	normal	Based on DHV Consultants, 2002; Seyoum, 2008
O&M pit latrine (including emptying)	ETB y^{-1}	250±20	normal	Based on DHV Consultants, 2002; Seyoum, 2008
Lifespan pit latrine	y	10±2	normal	Assumption
Investment UDDT	ETB	2,000±500	normal	Based on AMU and ARB, 2007; Seyoum, 2008
O&M UDDT (including emptying)	ETB y^{-1}	400±40	normal	Based on Seyoum, 2008
Lifespan UDDT	y	15±2	normal	Assumption
Investment septic tank (4 households)	ETB	9000±500	normal	DHV Consultants, 2002
O&M septic tank	ETB y^{-1}	1,400±100	normal	DHV Consultants, 2002
Lifespan septic tank	y	15±2	normal	Assumption
Investment anaerobic digester (4 households)	ETB	14,000±1000	normal	Based on Seyoum, 2008
O&M anaerobic digester	ETB y^{-1}	2,000±200	normal	Based on Seyoum, 2008
Lifespan anaerobic digester	y	15±2	normal	Assumption

F.6 Processes Markets and Livestock

Parameters used to calculate mass, nutrient, energy and cost flows of the Processes Markets and Livestock

Parameter	Unit	Average & Stdev	Distribution	Source
Manure in Arba Minch	t y^{-1}	182,000±20,000	normal	Based on Boxberger et al., 1995; Plückers, 2009
Organic waste in market as percentage of sold food	-	0.15±0.05	tnormal (0-1.0)	Assumption
Ratio of imported produce	-	0.12±0.02	tnormal (0-1.0)	FAO, 2009
Manure used in households	-	0.1±0.02	tnormal (0-1.0)	Estimation
N in cow manure	g_N kg^{-1}	4.5±0.5	normal	Stegmann, 2007
P in cow manure	g_P kg^{-1}	0.6±0.1	tnormal (0.2-1.0)	Stegmann, 2007
C in cow manure	g_{TOC} kg^{-1}	10.8±1	normal	Based on Gac et al., 2007
VS in cow manure	g_{VS} kg^{-1}	100±50	tnormal (0-1.0)	Smil, 1983

F.7 Process Collection/Transport

Parameters used to calculate energy and cost flows of the Process Transport

Parameter	Unit	Average & Stdev	Distribution	Source
Lorry capacity liquids collection	t	8±0	normal	Durbec, 2009
Fuel consumption liquids transport, average	l km^{-1}	0.33±0.03	normal	Durbec, 2009
Lorry capacity solid collection	t	5±0	normal	Assumption
Fuel consumption solids transport, average	l km^{-1}	0.20±0.02	normal	Assumption (see also Annex A.7)
Lower heating value fuel	kWh l^{-1}	8.9±0.3	normal	ORNL, 2008
Primary energy factor fuel	-	1.1±0.1	normal	DIN, 2007
Transport cost (lorry)	ETB km^{-1}	15±1	normal	Mindachew, 2009
Donkey cart cost	ETB y^{-1}	14,500±1,500	normal	Mindachew, 2009
Donkey cart capacity	t y^{-1}	800±200	normal	Mindachew, 2009

F.8 Process Treatment (co-composting & urine storage)

Parameters used to calculate mass, nutrient, energy and cost flows of the Process Treatment

Parameter	Unit	Average & Stdev	Distribution	Source
Sludge volume after dewatering	-	0.4±0.1	tnormal (0-1.0)	Strauss et al., 2003

Parameter	Unit	Average & Stdev	Distribution	Source
Compost per ton input	$t\ t^{-1}_{input}$	0.61±0.06	tnormal (0-1.0)	Belevi, 2002
N transfer coefficient to compost	-	0.69±0.2	tnormal (0-1.0)	Belevi, 2002
N transfer coefficient gaseous emissions	-	0.3±0.1	tnormal (0-1.0)	Belevi, 2002
P transfer coefficient to compost	-	0.99±0.1	tnormal (0-1.0)	Belevi, 2002
P transfer coefficient to leachate	-	0.01±0.01	tnormal (0-1.0)	Belevi, 2002
TOC transfer coefficient to compost	-	0.45±0.1	tnormal (0-1.0)	Leitzinger, 1999 451 /id}
TOC transfer coefficient gaseous emissions	-	0.45±0.1	tnormal (0-1.0)	Morand et al., 2005
Cost urine storage	ETB m^{-3}	1,700±100	normal	based on market prices
Lifespan urine storage	y	15±1	nomal	assumption
O&M urine storage	-	0.02±0.002	tlognormal (0-1.0)	assumption
Labour costs urine storage	ETB y^{-1}	30,000±5,000	normal	based on Mindachew, 2009
Benefit compost	ETB kg^{-1}	1.5±0.2	normal	Plückers, 2009
Benefit nitrogen	ETB kg_N^{-1}	12.0±1.2	normal	own calculation based on
Benefit phosphorus	ETB kg_P^{-1}	25.0±5.0	normal	market prices for DAP and urea

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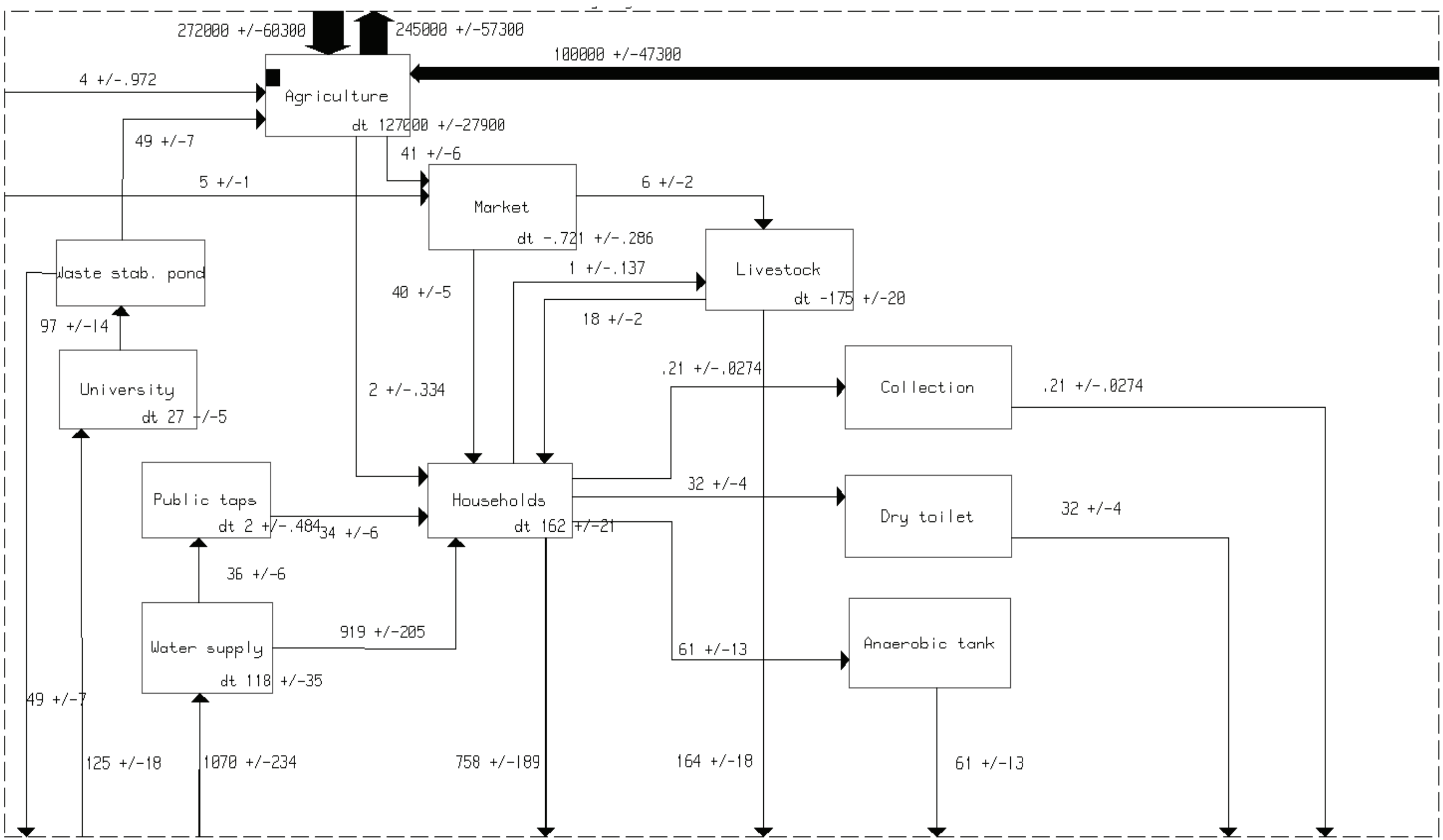
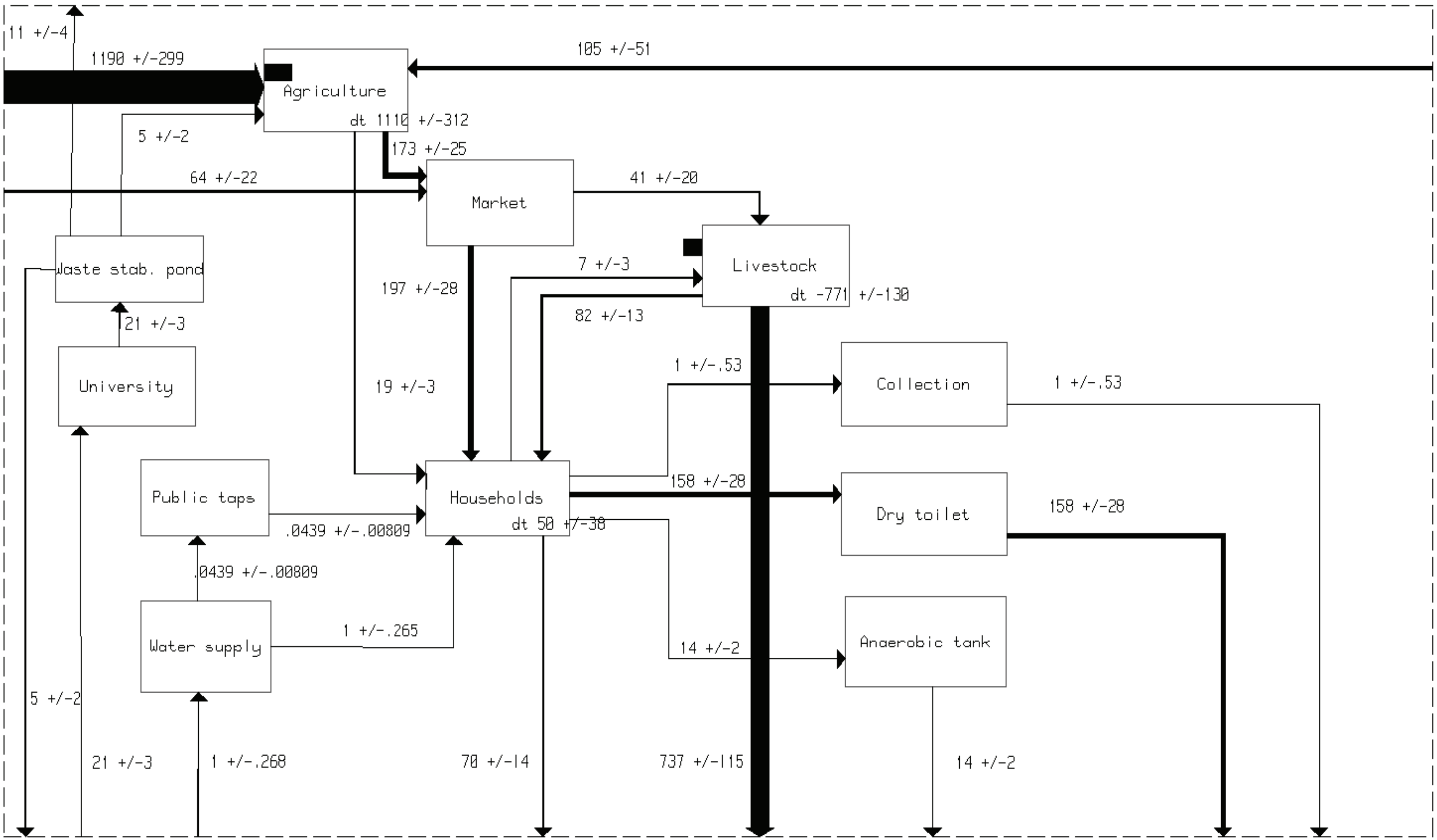
Annex G: Flow schemes of system 1 CuSit (Arba Minch)

The following pages exemplarily illustrate the flow schemes for the different modelled substances, i.e. mass flows, nitrogen flows and phosphorus flows for system 1 CuSit. The flow schemes of the other systems of the case study Arba Minch can be requested at ceMFA@ymail.com. The general system layout including the names of all processes and flows are depicted in detail in Section 3.6.2.

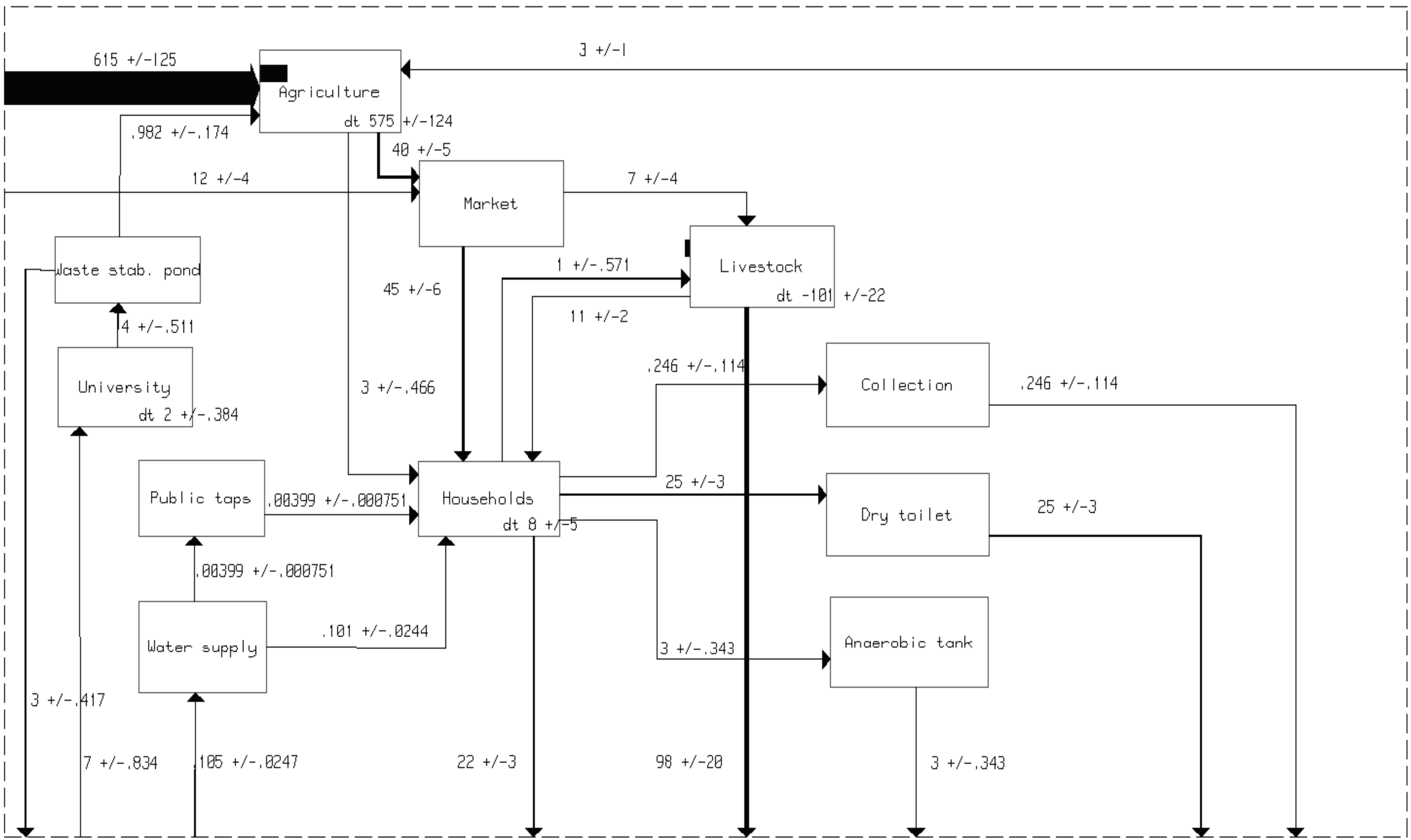
1 CuSit - Mass flows [1000 t y^{-1}]



1 CuSit - Nitrogen flows [t y^{-1}]



1 CuSit - Phosphorus flows [t y^{-1}]



Annex H: Sensitivities of the Arba Minch systems

Annex G shows the key parameters (up to ten are included) for each of the criteria nutrient recovery, energy demand per person and costs per person. The relative sensitivity of every parameter is shown in percent, i.e. a parameter change of 10% results in a variable change of the shown percentage. A negative value indicates that if the value of the parameter is reduced, then the value of the variable increases. Only relative sensitivities greater than 1% or smaller than -1% are included.

2 CoComp	Rate	3 UDDT	Rate	4 AnDig	Rate
Nitrogen Recovery (values indicate the percentage of variable change at 10% parameter change)					
Inhabitants of Arba Minch	10.0	Inhabitants of Arba Minch	10.0	Transfer coefficient N to compost	10.0
Transfer coefficient N to compost	10.0	N in excreta	6.5	Ratio of slurry added to anaerobic digestion	8.3
Ratio of pit sludge added to co-composting	5.0	Rate of urine collection in UDDT	6.1	N in manure	7.5
TS in organic waste	4.8	Ratio of UDDT implementation	5.6	Manure production in Arba Minch	7.4
N in organic waste	4.8	Transfer coefficient N to compost	5.7	Ratio of manure added to anaerobic digestion	7.4
Ratio of org. waste from market added to co-composting	3.6	TS in organic waste	2.0	Inhabitants of Arba Minch	2.6
Organic waste in market	3.6	N in organic waste	2.0	Ratio of org. waste (market) added to co-composting	1.7
N in manure	2.8	Ratio of org. waste (market) added to co-composting	1.5	N in organic waste	1.7
Amount of faeces per person	2.7	Organic waste in market	1.5	TS in organic waste	1.7
N in excreta	2.4	N in manure	1.4		
Phosphorus Recovery (values indicate the percentage of variable change at 10% parameter change)					
Inhabitants of Arba Minch	10.0	Inhabitants of Arba Minch	10.0	Transfer coefficient P to compost	10.0
Transfer coefficient P to compost	10.0	Transfer coefficient P to compost	6.6	Ratio of slurry added to anaerobic digestion	8.2
Ratio of pit sludge added to co-composting	5.3	P in excreta	6.3	P in manure	6.0
TS in organic waste	4.5	Ratio of UDDT implementation	3.6	Manure production in Arba Minch	5.9
P in organic waste	4.5	Rate of urine collection in UDDT	3.4	Ratio of manure added to anaerobic digestion	5.9
P in excreta	3.5	P in organic waste	2.4	Inhabitants of Arba Minch	4.1
Transfer coefficient P to pit sludge	3.3	TS in organic waste	2.4	Ratio of anaerobic treatment	2.2

2 CoComp	Rate	3 UDDT	Rate	4 AnDig	Rate
Ratio of org. waste (market) added to co-composting	3.3	Ratio of org. waste (market) added to co-composting	1.8	Transfer coefficient P to faecal sludge	2.2
Organic waste in market	3.3	Organic waste in market	1.8	Ratio of org. waste (market) added to co-composting	1.8
P in manure	2.0	P in manure	1.3	Organic waste in market	1.8
Cost per person (values indicate the percentage of variable change at 10% parameter change)					
O & M pit latrine	3.5	O & M UDDT	3.2	Ratio of slurry added to anaerobic digestion	6.9
Investment cost pit latrine	2.2	Amount of urine per person	2.5	Volume reduction in anaerobic plant	6.9
Cost of drinking water	1.8	Rate of urine collection in UDDT	2.5	Rate of anaerobic plants	6.3
Water consumption	1.5	Transport cost per km	2.0	Water for toilet flushing	4.8
Amount of faeces per person	1.4	Maximum distance household	1.5	Transport cost per km	3.5
Ratio of pit sludge added to co-composting	1.4	Rate of UDDT implementation	1.3	Maximum distance household	3.5
Transport cost per km	1.1	Investment cost UDDT	1.2	O&M cost biogas plant	2.2
Ratio of open defecation	-1.5	Lifespan UDDT	-1.1	Household size	-2.6
Lifespan pit latrine	-2.1	Truck capacity	-2.0	Capacity of vacuum truck	-3.5
Household size	-6.3	Household size	-5.6	Rate of biogas plants among anaerobic plants	-5.2
Energy per person (values indicate the percentage of variable change at 10% parameter change)					
LHV charcoal	5.7	LHV charcoal	5.8	LHV charcoal	9.1
Charcoal use per person	5.7	Charcoal use per person	5.8	Charcoal use per person	9.1
LHV fuelwood	2.9	LHV fuelwood	2.9	Stove efficiency	6.2
Fuelwood use per person	2.9	Fuelwood use per person	2.9	LHV fuelwood	4.5
Food intake per person	1.3	Food intake per person	1.3	Fuelwood use per person	4.5
Agricultural yield	1.3	Agricultural yield	1.3	Manure added to anaerobic digestion	-4.2
N fertiliser application rate	1.2	N fertiliser application rate	1.2	Biogas efficiency	-6.2
Energy for N fertiliser production	1.1	Energy for N fertiliser production	1.0	Biogas production per VS	-6.2
				Energy yield from biogas	-6.2
				Rate of biogas plants among anaerobic plants	-6.4

Curriculum Vitae

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