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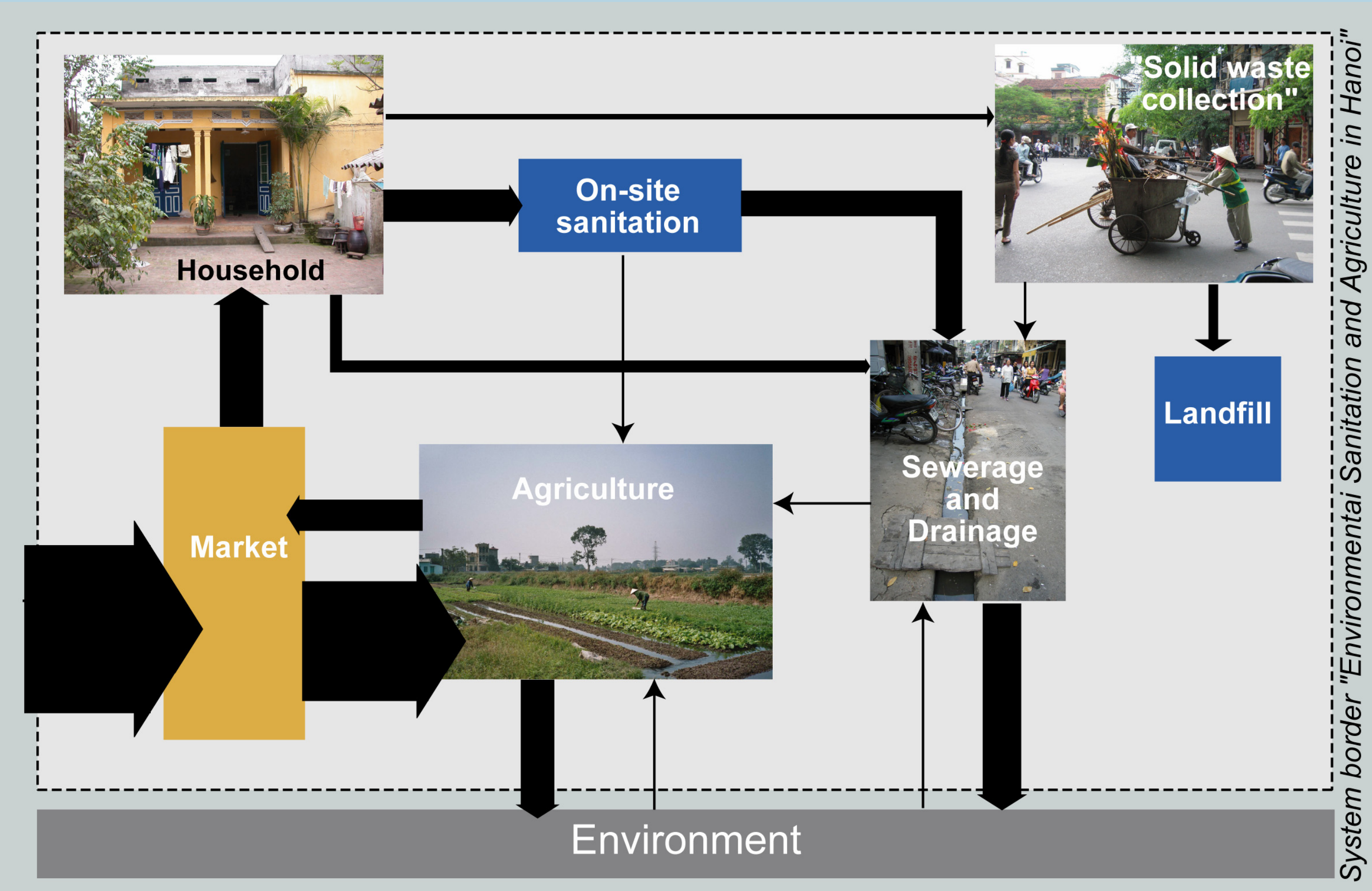
Material Flow Analysis

A Tool to Assess Material Flows for Environmental Sanitation Planning in Developing Countries

Agnès Montangero



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Foreword

Environmental protection and conservation of limited non-renewable resources are important aspects to consider when planning sustainable environmental sanitation systems. The material flow analysis (MFA) method studies the fluxes of resources used and transformed as they flow through a region. It can thus be used to optimize resource management when planning environmental sanitation systems. Environmental sanitation systems comprise water supply, sanitation, drainage and solid waste management.

MFA fits well into the Household-Centered Environmental Sanitation (HCES) planning approach characterized by a “circular model, which emphasizes resource conservation and reuse to reduce waste disposal, in place of the traditional linear model of unrestricted supply and subsequent disposal” (Eawag, 2005). The present publication attempts to provide advices to environmental sanitation planners, particularly those implementing the HCES approach, on how to assess material flows with limited data availability.

This publication is based on a case study conducted in Hanoi, Vietnam by Sandec and its partners from 2004 to 2006. The case study results have been published in journals and in a PhD thesis report on which the present publication is based:

- Montangero A, Belevi H (2008). An approach to optimise nutrient management in environmental sanitation systems despite limited data. *Journal of Environmental Management* 2008; 1538–1551.

- Montangero A, Belevi H (2007). Assessing nutrient flows in septic tanks by eliciting expert judgement: A promising method in the context of developing countries. *Water Research* 2007; 41: 1052-1064.

- Montangero A, Cau LN, Viet Anh N, Tuan VD, Nga PT, Belevi H. (2007). Optimising water and phosphorus management in the environmental sanitation system of Hanoi, Vietnam. *Science of the Total Environment*; 384: 55–66

- Montangero, A (2006). Material flow analysis for environmental sanitation planning in developing countries: an approach to assess material flows with limited data availability. PhD thesis University Innsbruck.

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We hope that this report will provide a valuable tool for environmental sanitation planners in low and middle income countries who wish to bring in the resource management perspective.

Dübendorf, August 2007 A. Montangero

Content

1. Introduction

1.1 Conventional sanitation system: linear water and nutrient flows

Clean water as excreta carrier

Water, nutrient and energy management practices of the conventional sanitation system are unsustainable (Fig. 1). Large quantities of clean, potable water are used to carry excreta through sewer systems even though water scarcity affects many regions of the world. The world's drinking water resources decreased from 17,000 m^3 $capita^{-1}$ in 1950 to 7,000 in the 1990s (UNDP, 1998) as a result of decreasing freshwater quantity and a near doubling of the world's population (Berndtsson and Hyvönen, 2002).

Fish dying

Moreover, some 90% of the wastewater generated worldwide are reportedly discharged untreated into receiving water bodies (GTZ, 2006). Excessive nutrient loads in aquatic systems stimulate plant growth and reduce dissolved oxygen in the water (eutrophication). This, in turn, can have adverse impacts such as fish dying, loss of aesthetic and recreational value, and high cost of treating raw drinking water (UNEP, 2002).

The phosphorus crisis

On the other hand, nutrients contained in wastewater discharged to the environment are "lost" and artificial fertilizers must be produced to cover the agricultural nutrient demand. Production of artificial nitrogen fertilizer is energy-intensive and phosphate rocks are mined for the production of phosphorus fertilizers. However, phosphorus reserves are likely to be depleted within 50-100 years (Cordell, 2005).

High costs and energy requirement to remove valuable nutrients

Although a large fraction of nutrients contained in wastewater is removed in wastewater treatment plants equipped with tertiary treatment, it can generally not be reused in agriculture. The nitrogen removal process converts ammonia to nitrate and subsequently to nitrogen gas, it is therefore lost to the air. Nowadays this is the most costly part of wastewater treatment plants (Maurer et al., 2002). Besides, the same amount of energy required for removing 1 kg of nitrogen in a wastewater treatment plant is needed to produce 1 kg of nitrogen fertilizer (45 MJ $kg^{-1}{}_{N}$, Maurer et al., 2003). The phosphorus removal process allows its accumulation in the treatment plant sludge. However, as sludge also contains hazardous substances such as heavy metals, its application in agriculture represents a risk and is expected not to be allowed any more in Europe in the near future (Maurer et al., 2002).

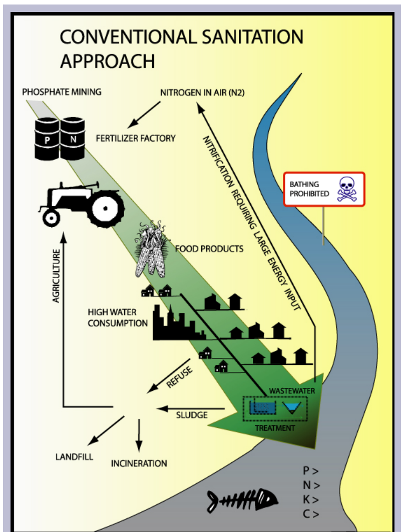


Figure 1: Conventional sanitation approach (source: Morel, A; adapted from Otterpohl R. http://www.otterwasser.de)

There is a need for new environmental sanitation concepts that contribute to protecting the natural environment and to conserving resources.

1.2 Material flow analysis as a tool for closing the water and nutrient cycle

What is material flow analysis?

The material flow analysis (MFA) method studies the fluxes of resources used and transformed as they flow through a region. In industrialized countries, MFA proved to be a suitable instrument for the early recognition of environmental problems and development of countermeasures (Baccini and Brunner, 1991). It is a widespread tool in many fields such as waste and wastewater treatment, agricultural nutrient management, water quality management, resource conservation and recovery (Brunner and Rechberger, 2004). It can be applied for example to evaluate the impact of changes in consumption patterns, solid waste and wastewater treatment infrastructure, peri-urban agricultural production, as well as waste and wastewater reuse practices on resource consumption and environmental pollution. It provides useful information that can be used by planners and decision-makers to optimize resource management in the environmental sanitation system for example by “closing the water and nutrient loop” (Fig. 2).

MFA helps optimize resource management

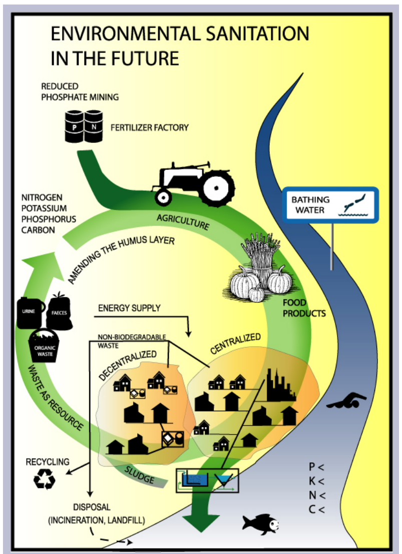


Figure 2: Environmental sanitation in the future (source: Morel, ????; adapted from Otterpohl R. http://www.otterwasser.de)

Data scarcity restrains the use of MFA in developing countries

However, limited data availability, reliability and collection means (available laboratory equipment, trained laboratory staff, financial and human resources) are common problems faced by developing countries restraining the use of MFA as a policy-making tool.

1.3 What is the purpose of this publication?

In order to facilitate the application of MFA as part of the environmental sanitation planning process in developing countries, this publication aims at providing practical recommendations on how to assess material flows despite limited data availability.

This report is intended for water and environmental sanitation professionals with a natural science or engineering background and interested to bring in the resource management perspective in environmental sanitation planning at city or regional level.

The publication attempts to provide advices on how to assess material flows with limited data availability on the basis of a case study conducted in Hanoi, Vietnam. Basic principles of the MFA method are presented in Chapter 2 and are followed by a detailed description of the procedure proposed to optimize water and nutrient flows in a regional environmental sanitation system (Chapter 3)..

2. Material flow analysis: basic principles

The material flow analysis method is described in Baccini and Brunner (1991), Baccini and Bader (1996) and Brunner and Rechberger (2004). Material flow analysis comprises the following main steps:

The main steps of an MFA

1. System analysis (selection of the relevant goods, processes, indicator substances, and system boundaries).
2. Quantification of mass flows of goods and of indicator substances.
3. Schematic representation and interpretation of the results.

Substance (or material): a chemical element such as nitrogen and phosphorus or a chemical compound such as CO_2 and NH_3.

Good: a substance or a mixture of substances with a function valued by man such as food, solid waste and wastewater.

Process: transformation, transport or storage of goods and substances such as household, wastewater treatment plant, agricultural soils.

The phosphorus crisis: zooming on the households

These steps are illustrated by the following example: what are the main phosphorus flows in a typical household in Hanoi, Vietnam? What are the main sources of phosphorus? And how could phosphorus use and discharge into the environment be reduced?

1. System analysis

As a general rule, the system should be geared to specific objectives or questions such as the ones mentioned in the last paragraph. In this example, the indicator **substance** is clearly phosphorus. Relevant **goods** are the input and output goods to and from the household, which contain phosphorus: food and detergent, excreta, blackwater (toilet wastewater), greywater (domestic wastewater from kitchen, laundry, bathroom, etc. except toilet wastewater), and kitchen waste.

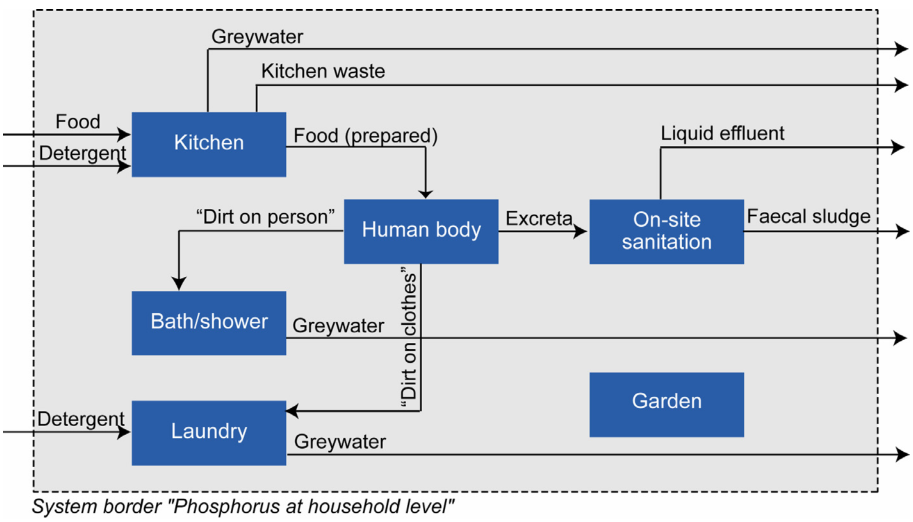


Figure 3 System comprising relevant goods and processes used to describe phosphorus flows at household level

In order to get a detailed overview of phosphorus pathways at household level, the following **processes** are selected: kitchen (cooking), human body (food consumption, digestion, and excretion), toilet and on-site sanitation system, bath/shower, and laundry. Phosphorus containing goods are being transformed in these processes. For example, phosphorus enters the human body as food and leaves it, after digestion, with the excreta. The **system boundary** includes all household activities. The system is represented in Figure 3: The boxes designate processes and the arrows represent the flows.

2. Quantification of flows

The second step consists in quantifying good and substance flows in the system. Only the substance (phosphorus) flows are of interest in this case. Substance flows are often calculated based on the mass flows of goods and substance concentrations in these goods. For example, the phosphorus flow in laundry detergent can be assessed by multiplying the detergent consumption with the phosphorus content in detergent. However, information on substance flows may also be directly available. Examples on how to assess good or substance flows are given in Chapter 3.

3. Schematic representation and interpretation

In the third step, the assessed good and/or substance flows are represented graphically and interpreted (Fig. 4). A closer look at the figures shows that the total input (1.6 g P cap^{-1} day^{-1}) does not exactly correspond to the total output (1.7 g P cap^{-1} day^{-1}). This is due to the fact that each flow has a certain uncertainty that must be taken into account. Chapter 3 gives examples how to consider uncertainty. According to Brunner and Rechberger (2004), it is very rare that a balance between inputs and outputs of a measured system yields an error less than 10% of the total flow.

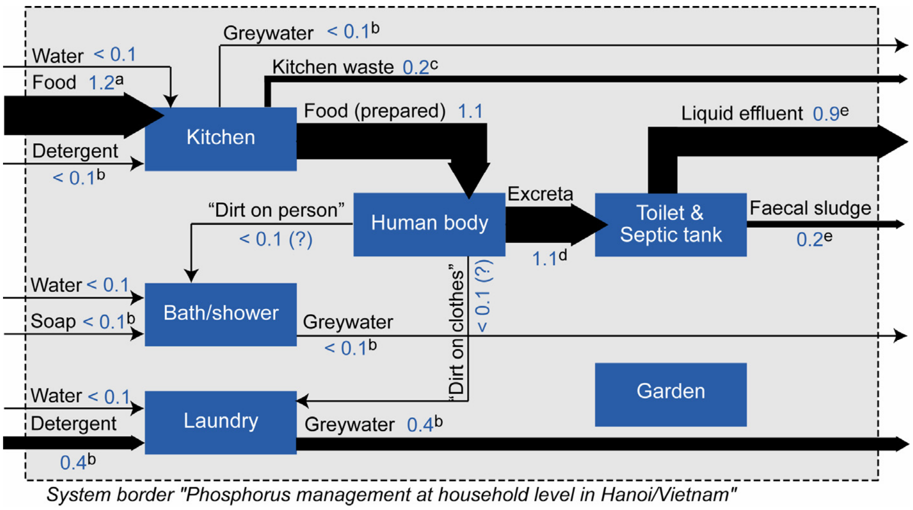


Figure 4 Average phosphorus flows in urban household in Hanoi. Flows are expressed in g P cap^{-1} day^{-1}. Data are based on: [a] FAO (1972), FAOSTAT (2004). [b] Büsser (2006). [c] Diaz et al. (1996), Rytz (2001), Schouw et al. (2002b), Strauss et al. (2003), Sinsupan (2004). [d] Polprasert et al. (1981), Heinss et al. (1998). [e] Montangero and Belevi (2007). Other flows are based on assumptions.

As aforementioned, global phosphorus reserves are limited and the question therefore arises how its consumption can be lowered. The main phosphorus sources in Hanoi are the food and laundry detergent. According to Büsser (2006), phosphorous is also used in detergents for dishwashing machines, but these are not common in Hanoi yet. Obviously, phosphorus in food supply is essential. However, banning phosphorus-containing detergent would not only reduce household P consumption by a quarter but also decrease the P load in greywater and thus lower the amount of P discharged into the environment or to be removed from wastewater in a wastewater treatment plant.

Moreover, reusing part of the phosphorus contained in the output goods (liquid effluent and sludge from septic tanks, greywater and kitchen waste) for irrigating/fertilizing the garden would further reduce P consumption as fertilizer and lower the amount of P discharged into the environment. It must be noted that the type of on-site sanitation facility strongly influences the characteristics of its output goods. Urine diversion latrines, for example, lend themselves better to nutrient recovery than septic tanks (Chapter 3.5.2).

The HCES approach recommends that "solutions of environmental sanitation problems should take place as close as possible to the place where they occur" so as to avoid "exporting" the problems (Eawag, 2005). In this sense, zooming in on the "household" process enables to develop household-level measures which could allow to lower wastewater treatment requirement at a "higher" level, for example at city level and avoid causing nuisance to downstream communities by reducing pollutant load into the environment.

However, after having zoomed in the household process, it might be useful to zoom out and get an overview of phosphorus flows at a regional level. Even though, for example phosphorus in food supply is a "given value", the type of food production has a strong impact on phosphorus fertilizer consumption (Chapter 3.5.3). Chapter 3 will thus describe how to assess material flows in a regional environmental sanitation system exemplified by the case of Hanoi province in Vietnam.

3. Optimizing water and nutrient flows in an environmental sanitation system - The example of Hanoi province in Vietnam

An approach to assess material flows within the environmental sanitation system of a region with limited data availability has been developed (Fig. 5). It is based on the method of material flow analysis (Baccini and Brunner, 1991) and attempts to integrate ways to deal with scarce data in order to render the method more affordable and to avoid drawing wrong conclusions.

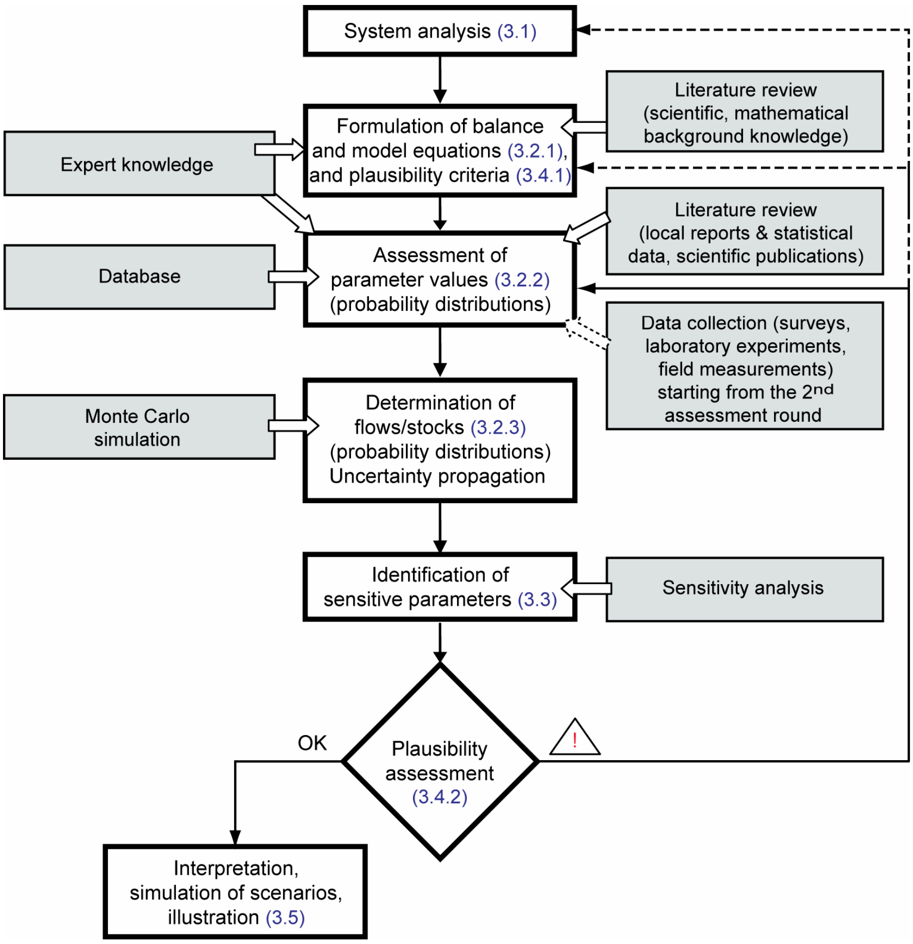


Figure 5 Proposed procedure to assess material flows despite limited data availability and reliability. The different steps are described in the chapters below (chapter numbers in brackets).

The different steps of the approach illustrated in Fig. 5 are discussed and exemplified in Chapters 3.1-3.5 on the basis of a water and nutrient management case study conducted in Hanoi province, Vietnam.

3.1 System analysis

In this step, the relevant goods, processes, indicator substances and system boundaries (spatial and temporal) are selected. As mentioned in Chapter 2, the system should be geared to analyze specific issues, answer specific questions. In the case of Hanoi, the system was developed based on the following issues.

Selecting key issues

- Hanoi's water supply relies mainly on groundwater; however, the city's water demand is already reaching the aquifer's recharge rate. The groundwater level is lowered by overexploitation and sectors of the city are suffering from land subsidence. Water demand is likely to increase in the coming decades due to population growth, improved living standards and industrial development.

- Lakes, ponds and canals in Hanoi are seriously affected by untreated domestic and industrial wastewater. They exhibit very low dissolved oxygen level and high concentrations of organic matter, nutrient, and pathogenic microorganisms (CEETIA, 2002). This causes a threat to health and the environment.

- Peri-urban agriculture in Hanoi is of key importance in the supply of food and provision of income to the poorest section of the population. The amount of land available for agriculture is decreasing due to rapid urbanization, causing an increase in land prices and forcing farmers to intensify their land use. Farmers tend to use more fertilizers in an attempt to enhance yield and benefit from their decreasing land area.

It was decided that the system should support the analysis of measures aiming at reducing water consumption, nutrient discharge into surface water and fertilizer use.

Understanding the environmental sanitation system

A first qualitative description of the system was conducted on the basis of discussions with local specialists, field visits, and review of relevant literature (Box 1). Brunner and Rechberger (2004) state that the definition of the system is a decisive and demanding task. Poor results of MFA can often be traced back to an unsuitable system definition. The system analysis should therefore be given particular care and be critically revised during the later steps of the MFA.

Box 1 Environmental sanitation in Hanoi

Water supply

Hanoi's water supply mainly relies on groundwater. 8 water treatment plants and a number of small water stations using groundwater from a confined aquifer supply water to the population living in the urban districts (Nga Tran Thi Viet, 2005). Besides, a large number of wells are exploited by factories and households in peri-urban districts (Hanoi DOSTE, 2003).

Sanitation and drainage

Hanoi's inhabitants rely on on-site sanitation such as septic tanks, dry single and double pit latrines with or without urine diversion, pour flush latrines, biogas latrines and bucket latrines. Septic tank is the most common on-site sanitation option in the urban area. As groundwater table in Hanoi is high, most septic tanks are not equipped with a leaching pit or field but the septic tank effluent flows in the sewerage and drainage system.

Wastewater flowing through a street drain in Hanoi (© Ruth Scheidegger, Eawag).

As septic tanks are only rarely emptied, solids are carried over and accumulate in the sewerage and drainage network, increasing the pollutant load in surface water and reducing the capacity of the drainage network. Greywater usually flows untreated to the drainage network.

Drainage and sewerage form a combined system (transport of domestic, industrial and storm water) that flows by gravity into lakes, ponds and rivers (Wnukowska, 2004). Except for domestic black wastewater which is treated in septic tanks and a limited amount of industrial and hospital wastewater which passes through preliminary treatment, all wastewaters are discharged untreated into surface water bodies which drain towards the Nhue and Red rivers (Viet Anh et al., 2004).

Fecal sludge from septic tanks as well as pit latrines in the urban center is usually disposed of in a landfill. A small fraction of the sludge is co-composted with solid waste at the Cau Dien co-composting plant using forced aeration by blowers. Additives are used to increase the N, P, K content of the compost (Thai, 2005). In peri-urban areas, latrines are mainly emptied by individuals and their content is reused as fertilizer in agriculture or aquaculture either directly or after storage or on-farm composting with other wastes.

Solid waste management

A large percentage of the solid waste generated in the urban districts are collected and landfilled. A small proportion of the total waste volume (mainly market waste) is co-composted at the Cau Dien composting plant (Thai, 2005). The uncollected waste ends up on open dumps or in the drainage channels, is burnt in the open or recycled. In peri-urban areas, solid waste is mainly disposed of in open dumps, some is mixed with manure or other wastes and stored before being reused as fertilizer; food waste is sometimes fed to livestock. The large amount of solid waste disposed of in open dumps or discharged in open drainage channels also leads to pollution of water bodies.

Solid waste collection in Hanoi (© Ruth Scheidegger, Eawag).

Industrial waste is usually collected and disposed of with domestic solid waste at the Nam Son landfill or in open dumps. The Nam Son landfill also includes a hazardous waste treatment plant. Only a small number of larger, foreign-invested firms have hazardous waste treatment systems, and some have agreements with suppliers to return hazardous waste back to the manufacturer for treatment and disposal (Palladino, 2001).

The Nam Son landfill includes refuse cells, synthetic landfill liners (plastic liner) placed on a clay layer at the bottom of the landfill, and embankments to prevent the infiltration of leachate. The collected leachate is treated in a series of 3 ponds. The cells are covered with soil once they are filled with solid wastes (Hanoi URENCO, 2004).

Agriculture

Cereals, vegetables and pulses are the major crops grown in Hanoi (Anh et al., 2004). The urbanization process has various impacts on the peri-urban agricultural sector. First, urban growth decreases the amount of land available for agriculture, causing an increase in land prices and forcing farmers to intensify their land use. Second, the demand for food products such as meat (pork), vegetables, fruit and fish increases (van den Berg et al., 2003).

Peri-urban Hanoi: Mix of agricultural and new housing zones.

Urban and industrial effluents are used by the farmers in Thanh Tri (South of Hanoi) as a free substitute for chemical fertilizers and fish food (van den Berg et al., 2003). Wastewater-fed fish ponds and wastewater irrigated paddy fields reportedly give higher yields and higher financial benefits (Nguyen Ngoc Thu, 2001). However, wastewater reuse is also associated with risks.

Wastewater-irrigated fields in Thanh Tri, Hanoi.

Identifying appropriate indicator substances and relevant processes and goods

On the basis of the selected key issues, the indicator substances phosphorus, nitrogen and water have been selected. The system should include both urban environmental sanitation and peri-urban agriculture as reuse of treated wastewater and organic solid waste as water and nutrient sources for food production in the peri-urban area could reduce both resource consumption (water, energy, phosphate rock) and nutrient discharge into the environment. Furthermore, the links between the environmental sanitation and peri-urban agricultural system and the environment should also be identified.

The households play a key role in the system. Water consumption behaviour, selection of water-efficient house installations and reuse of rainwater or greywater for toilet flushing or garden irrigation are some of the factors affecting water exploitation. Moreover, the type of on-site sanitation installations used also influences the amount of nutrients available for use as fertilizer.

The system should thus comprise the following elements:

- The **household** as water, detergent and food consumer as well as excreta, wastewater and organic waste producer.
- The **urban environmental sanitation sector**: water supply, on-site sanitation, sewerage & drainage network, solid waste collection, landfill, open dumps, and composting.
- The **peri-urban food production sector**: crop production, livestock production and aquaculture.
- The **environment** as water supply source (groundwater, surface water, atmosphere) as well as nutrient supplier (nitrogen fixation, nutrient deposition from the atmosphere) and receiver (surface water, soil, groundwater, and atmosphere).

The system proposed is illustrated in a simplified way in Fig. 6 and in more detail in Fig. 7. The boxes designate processes and the arrows represent flows. The simplified system illustrated in Fig. 6 is appropriate to give a rapid overview over the main processes and flows; while the system described in Fig. 7 is a good basis for a more comprehensive analysis of the impact of changes (e.g. household consumption patterns, type of on-site sanitation infrastructure, wastewater and solid waste reuse strategies, type of crops and livestock categories, etc.) on water and fertilizer consumption as well as on nutrient discharge into the environment.

The process "market" contained in Fig. 7 represents a platform for the exchange of goods produced in Hanoi and distributed to the households in and outside Hanoi province. It is also the place where "imported" products such as food and agricultural inputs (fertilizer) are distributed to the households and to agricultural processes. The process "industry" was left outside the system border; only industrial water consumption, solid waste and wastewater generation have been considered.

Some processes are divided into sub-processes. The different types of on-site sanitation systems operating in Hanoi (septic tank, single pit latrine, pour flush infiltration latrine, double pit urine diversion latrine, biogas latrine, bucket latrine) are sub-processes of the process “on-site sanitation”. Similarly, the process “agriculture” is divided into crop production (major crops grown), livestock production (main livestock categories reared) and aquaculture.

Representing the system comprising processes, goods flowing to and from the processes, and system boundary

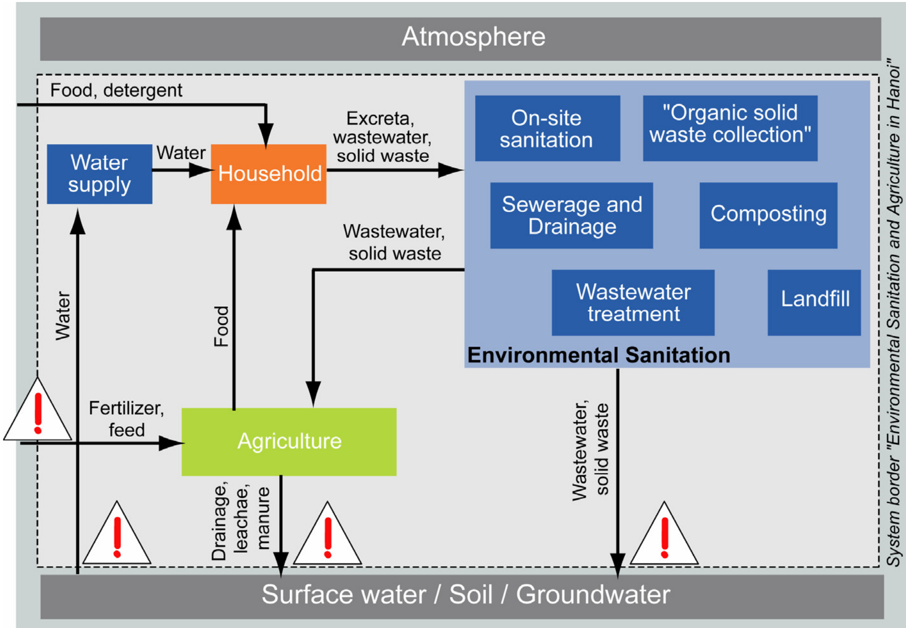


Figure 6 Simplified system representation of environmental sanitation and agriculture in Hanoi province. The exclamation marks indicate the aforementioned key issues. Only the main processes and goods are indicated here.

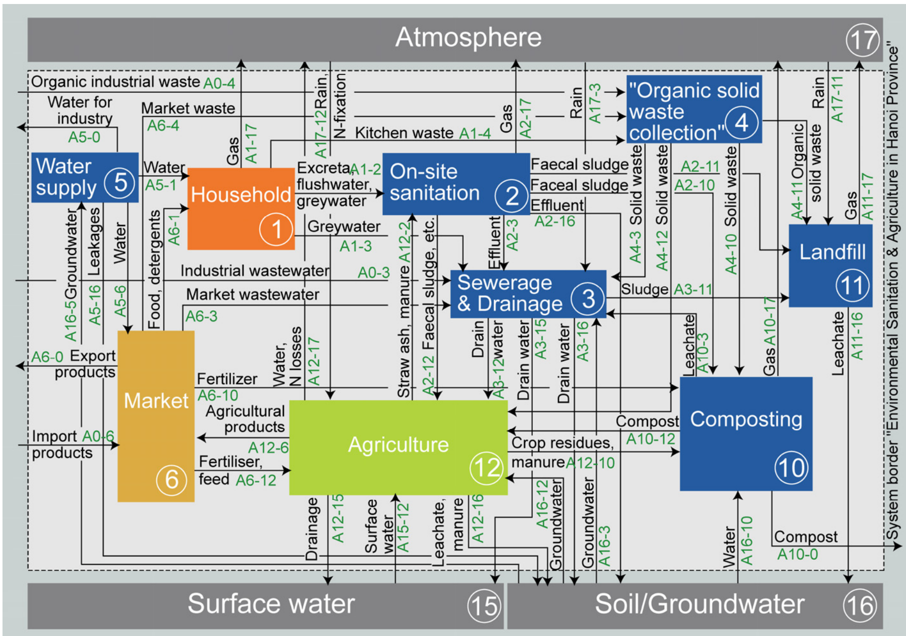


Figure 7 System description of environmental sanitation and agriculture in Hanoi province. Some process numbers are missing as some processes are not represented here.

The next steps will consist in identifying “control points”, i.e. factors in the different processes determining the key flows: groundwater withdrawal, nutrient use as fertilizer, and nutrient discharge into surface water. Moreover, the impact of different interventions at the “control points” on the key flows will be assessed. These steps can be supported by mathematical modelling of the material flow system (Box 2). Propositions for interventions should be developed in a multi-stakeholder process (see HCES approach: Eawag, 2005).

Box 2 Why and how to model material flow systems?

The mathematical description of MFA systems allows to simulate the impact of changes in the system and can therefore be used to evaluate potential environmental sanitation scenarios with regard to resource consumption and environmental pollution.

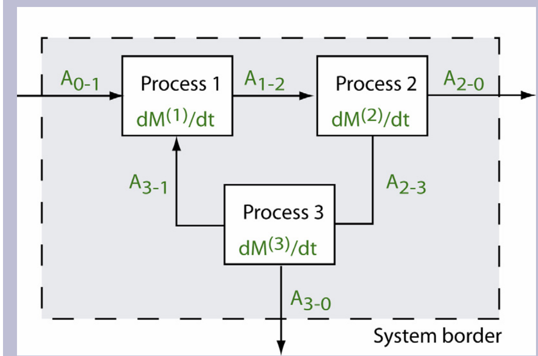
Development of a mathematical model for a specific MFA system consists of the following steps (Baccini and Bader, 1996):

1. **Identification of system variables** based on the system analysis.

The variables of an MFA system are:

- Mass (total mass or mass of substance i) increase or decrease in each process: $dM_i^{(j)}/dt$
- Mass flow (total mass or mass of substance i) from or to a process: $A_{i,j\text{-}s}$ or $A_{i,r\text{-}j}$

System analysis forms the conceptual model and basis of the mathematical model. The variables are determined by system analysis.



Example of a system analysis illustrating system variables (green).

An MFA system is regarded as fully described when all variables, i.e. stock change rates of goods or substances in each process and all good or substance flows in the system are known.

2. **Determination of the balance equations**

The law of mass conservation allows to formulate a balance equation for each process. Stock change rate, equaling input minus output flow, is defined for each process:

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where i expresses the indicator substance (for example P for phosphorus), j the process number, $M_i^{(j)}$ the stock of substance i in process j, t the time, r the source process, s the destination process, $A_{i,r\text{-}j}$ the input flow of substance i from the source process r to the process j, and $A_{i,j\text{-}s}$ the output flow of substance i from the process j to the destination process s. The left side of the equation indicates the stock change rate of substance i within the process j; the right side stands for the difference between input and output flows of substance i to and from process j. If i is not a chemical element but a compound, such as for example NO_3-, a production term (sources and sinks) should be displayed in the balance equation.

3. **Development of the model**

This step consists in determining "model equations" based on scientific and expert knowledge. They represent the characteristic features of the system and express how different parameters determine the variables in the system. For example:

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where p_1, p_2,..., p_n are parameters influencing the variable $A_{i,r\text{-}j}$. By formulating balance and model equations and assessing parameter values, material flows and stock change rates can be quantified.

3.2 Estimating material flows

3.2.1 Developing a material flow model

The system analysis forms the basis of the material flow model. Based on the system analysis, system variables are identified and system equations are formulated, whereby two types of equations can be distinguished: balance and model equations (Box 2). To establish model equations, a deep understanding of the phenomena occurring in each process and factors influencing material flows is needed. This understanding can build on expert knowledge (on the entire system or system parts as well as on specific processes), literature review or surveys and field measurements. It is recommended to develop the model **process by process**.

The following example illustrates how a simple model describing **phosphorus flows in the household process** in Hanoi can be developed. This process includes all household activities related to water and nutrient flows such as cooking, eating, defecating, bathing, washing dishes and clothes. Sub-processes are already described in Chapter 2. However, unlike in Chapter 2, "On-site sanitation" is not considered as a sub-process of the "household" but as a distinct process (see Fig. 10). This allows a finer analysis of the impact of different types of on-site sanitation installations on nutrient flows.

Understanding the phenomena occurring in the process

Phosphorus (P) inputs in the "household" are food and detergents. The food is consumed or ends up as food waste (inedible part or food leftovers). Some of it is flushed out during dish washing (greywater). P in detergents or cleansing products also ends up in the greywater. Consumed phosphorus leaves the human body with the excreta. In the case of Hanoi, mainly blackwater (excreta and flushwater) is discharged into on-site sanitation systems, while greywater flows directly into the sewerage and drainage network. Greywater is sometimes also pretreated with blackwater in septic tanks.

Identifying system variables based on the system analysis

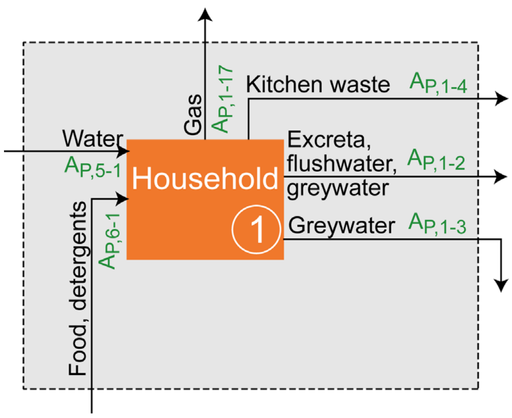


Figure 8 The household process in Hanoi with its input and output flows. See Fig. 7 for a complete picture of the system.

The **system variables** are the stock change rate of phosphorus in this process: $dM_P^{(1)}/dt$ and the input and output phosphorus flows: $A_{P,5-1}$, $A_{P,6-1}$, $A_{P,1-2}$, $A_{P,1-3}$, $A_{P,1-4}$, $A_{P,1-17}$ (see Figs. 7 and 8).

Basically, seven equations must be formulated in order to assess the seven variables.

Formulating balance and model equations

The **balance equation** for phosphorus in the household (HH) process can be formulated as follows:

$$\frac{dM_P^{(1)}}{dt} = A_{P,5-1} + A_{P,6-1} - A_{P,1-2} - A_{P,1-3} - A_{P,1-4} - A_{P,1-17}$$ *HH balance equation*

Six **model equations** are subsequently formulated. They represent the characteristic features of the system and express how different parameters determine the variables in the system:

The P flow from the process "market" to the process "household" is expressed as a function of the per capita food consumption and use of detergent (a_{food_f}, a_{det_d}) in kg cap^{-1} year^{-1}, P content in food items and P content in detergent ($C_{P,food\_f}$, and C_{P,det_d}) in g kg^{-1} and the number of inhabitants (n):

$$A_{P,6-1} = n \cdot (\sum_f a_{food\_f} \cdot C_{P,food\_f} + \sum_d a_{\det\_d} \cdot C_{P,\det\_d}) \cdot 10^{-3}$$ *HH model equation 1*

P input through water supply has been neglected in this model:

$$A_{P,5-1} = 0$$ *HH model equation 2*

The P flow from "household" to "on-site sanitation" systems is expressed as a function of the per capita P load in excreta ($a_{P,excreta}$), per capita P load in greywater ($a_{P,greywater}$), both in g cap^{-1} day^{-1}, fraction of greywater discharged into the on-site sanitation installations (r_{grey_ST}), and number of inhabitants:

$$A_{P,1-2} = (a_{P,excreta} + a_{P,greywater} \cdot r_{grey\_ST}) \cdot n \cdot 365 \cdot 10^{-3}$$ *HH model equation 3*

The P flow from "household" to "sewerage and drainage network" is expressed as a function of the per capita P flow in greywater, fraction of greywater discharged into "on-site sanitation" systems and number of inhabitants:

$$A_{P,1-3} = n \cdot a_{P,greywater} \cdot (1 - r_{grey\_ST}) \cdot 365 \cdot 10^{-3}$$ *HH model equation 4*

The P flow from "household" to "solid waste collection" is expressed as a function of the per capita P load in kitchen waste ($a_{P,kitchen\ waste}$) in g cap^{-1} day^{-1} and the number of inhabitants:

$$A_{P,1-4} = n \cdot a_{P,kitchen\ waste} \cdot 365 \cdot 10^{-3}$$ *HH model equation 5*

There is no P loss in gaseous form:

$$A_{P,1-17} = 0$$ *HH model equation 6*

Minimizing number of parameters and eliminating parameters difficult to assess

When developing model equations by limited data availability, it is important to minimize as much as possible the number of parameters. Moreover, equations containing parameters difficult to assess should be reformulated so as to eliminate these parameters.

This principle is illustrated by the example of the phosphorus load in human excreta. As the per capita P load in excreta is dependent on dietary habits, it will vary regionally. To avoid experimental determination of this load, the model was adapted so as to express it as a function of the total food protein (a_{TFP}) and vegetable food protein (a_{VFP}) supplied to the "household" in g cap^{-1} day^{-1} as well as of the ratio between P load in excreta and the sum of total and vegetable protein supply ($r_{P\ excreta}$) according to Jönsson et al. (2004):

$$a_{P,excreta} = r_{P\_excreta} \cdot (a_{TFP} + a_{VFP})$$

According to Jönsson et al (2004), "vegetable food stuffs contain on average twice as much phosphorus per gram of protein as animal ones". This is why the vegetable protein is counted twice in the above mentioned equation. Information on protein supply is available at country level (FAOSTAT, 2004). Moreover, food consumption survey data is available for several regions and cities. These surveys were conducted in Hanoi and provide data on food protein supply both in urban and peri-urban areas of Hanoi (Anh et al., 2004). If information on food consumption is not available, these surveys are more easily conducted than excreta characterisation. Jönsson et al. (2004) also provide a value for $r_{P\_excreta}$. The parameters $a_{P\_excreta}$ can thus be disregarded and "Household model equation 3" reformulated as follows:

$$A_{P,1-2} = (r_{P\_excreta} \cdot (a_{TFP} + a_{VFP}) + a_{P,greywater} \cdot r_{grey\_ST}) \cdot n \cdot 365 \cdot 10^{-3}$$ *HH model eq. 3*

3.2.2 Assessing parameter values

Balance and model equations have now been formulated. In order to quantify the variables (stock change rates and material flows), parameter values must be assessed.

Start by a rough parameter approximation

Only start conducting measurement campaigns once sensitive parameters have been identified

As illustrated in Fig. 5, **parameter assessment is an iterative procedure**. It is important to **start by a rough parameter assessment**, particularly where data collection means are limited. Parameter values should thus initially be assessed by reviewing local reports, statistical data, scientific publications and databases, and by eliciting expert judgement (Box 5). Variables are initially calculated (Chapter 3.2.3) on the basis of the first parameter approximation values. Plausibility of parameter values and model outcomes is subsequently assessed (Chapter 3.4). If not all the plausibility criteria are met, **the most sensitive parameters** (Chapter 3.3) **are reassessed more accurately**. A more differentiated literature review and/or field measurements or surveys can be carried out to obtain a more accurate assessment of the sensitive parameters.

This iterative procedure is very important as it **reduces the cost** of the flow estimation and hence of the entire material flow analysis. Why? Determining all model parameters experimentally (by sampling and analysis or surveys) would neither be feasible (cost, time, etc.) nor meaningful. It is difficult to determine which parameters should be assessed accurately at first sight and hence there is a risk to spend the limited budget on measuring few parameters, which may not significantly influence the key flows. One may thus end up having spent the entire budget but without a reasonable assessment of the most determining parameters. The iterative procedure should help avoiding ending up like that. The first rough parameter assessment serves as a basis to calculate material flows and identify the most sensitive parameters, i.e. the ones significantly influencing the key flows. The limited budget can now be spent for assessing the most sensitive parameters with a sufficient accuracy.

Large material flow uncertainties can lead to incorrect conclusions. Assuming that the phosphorus flow into surface water is estimated for

Consider data uncertainty

> Moreover, considering data **uncertainty** is of utmost importance. Parameters can be expressed as probability distributions instead of average values (Box 3). Parameter uncertainties are subsequently propagated through the model in order to assess uncertainty in material flows (Box 6).

the current situation as well as for a given scenario, comparison of both average flow values could lead to the conclusion that one of the flows is larger than the other. However, considering not only average values but also the ranges or uncertainties could lead to the conclusion that both flows are of the same order of magnitude. Working with probability distributions rather than mean values thus allows a more differentiated interpretation of the model outcomes.

First rough parameter assessment based on literature data or expert judgment

The iterative procedure and the consideration of uncertainty are illustrated here on the basis of the household parameters. Table 1 (Annex A) contains the results of the first rough parameter approximation. The parameters are not only characterized by an average value but by a distribution type and the characteristics of the distribution such as mean and standard deviation (Box 3). The number of inhabitants was for example assumed to be normally distributed as most values are likely to range around the mean value, with fewer values far below or far above the mean value. A mean value of 3.1 million was assessed on the basis of the values provided by the Hanoi Statistical Office (HSO, 2002) and assuming that 15% of Hanoi's inhabitants are unregistered (local expert knowledge). Standard deviation was assessed at 0.4 million based on the range of literature values.

More accurate assessment of the sensitive parameters based on a more differentiated literature review, surveys or field measurements

Variables were subsequently calculated on the basis of these parameter values (Chapter 3.2.3). However, several plausibility criteria could not be met (Chapter 3.4) and thus the sensitive parameters (identified in Chapter 3.3) should be reassessed more accurately. This reassessment is illustrated here based on one of the most sensitive parameters - the per capita P load in greywater. The first approximation, based on a general literature review, resulted in a very wide range of values: $a_{P,greywater}$ ranges between 0.1 and 3.5 with a mean value of 0.15 g cap^{-1} day^{-1}. Surveys and field measurements conducted in Hanoi resulted in a more accurate determination of detergent consumption, P content in detergent as well as greywater generation and P concentration in greywater (20 samples). The P load in Hanoi's greywater amounts to 0.5 (0.1-1.1) g cap^{-1} day^{-1} (0.6 in urban area and 0.4 in suburban area) (Büsser, 2006). Table 2

(Annex A) contains values of the reassessed parameters. Similarly, values for all sensitive model parameters were reassessed until all plausibility criteria were met.

Box 3 Uncertainty of model parameters

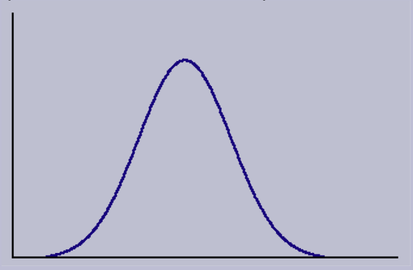
Considering data uncertainty is of utmost importance, particularly when data availability is limited. Not considering uncertainty could lead to incorrect conclusions, for example when comparing the impact of different scenarios. One way of characterizing uncertainty of model inputs (parameters) is to express them as probability distributions.

There are two distinct views of probability. The standard view of probability is the **frequentist view**: the probability of an event is derived from repeated experiments in which the frequency that an event occurs is analyzed. According to the **subjectivist** (or bayesian) **view** the "probability of an event is the degree of belief that a person has that it will occur, given all the relevant information currently known to that person. Thus the probability is a function not only of the event but of the state of information" (Morgan and Henrion, 1990).

In case of limited data availability, the type of probability distribution can usually not be determined from existing data but has to be assumed on the basis of our knowledge on each parameter (subjectivist view). According to Morgan and Henrion (1990), the **lognormal distribution** is appropriate to represent "physical quantities that are constrained to being non-negative, and are positively skewed, such as pollutant concentrations, stream flows", etc. Moreover, the lognormal distribution is particularly suitable to represent large uncertainties.

Example of a lognormal distribution.

The **normal distribution** provides a good model for a parameter, when there is a strong tendency for the parameter to take a central value, positive and negative deviations from this central value are equally likely, and the frequency of deviations falls off rapidly as the deviations become larger (StatSoft, Inc., 2006).



Example of a normal distribution.

According to Gränicher (1996), for the normal distribution, around 68% of the values lie within one standard deviation from the mean value and 95% within two standard deviations.

Use of the **uniform distribution** is appropriate when we are able to "identify a range of possible values, but unable to decide which values within this range are more likely to occur than others" (Morgan and Henrion, 1990).

Example of a uniform distribution.

Characteristics of the probability distributions, such as mean and standard deviation or minimum and maximum, were estimated based on knowledge on parameters, available data and reliability of the information sources or by applying **expert judgment elicitation technique** (Box 5).

Box 4 Probability “refresher”

How to build, represent and interpret probability distributions?

A **frequency histogram** can be built based on a series of data points as illustrated by the following example. We assume that n data points for the parameter “per capita household water consumption” are available:

Per capita household water consumption in Hanoi (n data points)

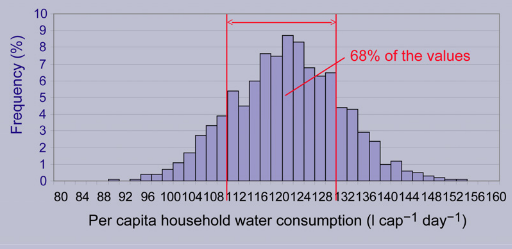
Household water consumption (l cap^{-1} day^{-1})	
Value 1	110
Value 2	127
Value 3	122
...	
Value n	120

These values can be classified in different intervals. The number of data points or percentage of data points contained in each interval can be plotted on a graph (frequency histogram).

Per capita household water consumption in Hanoi (frequencies of data points in different ranges)

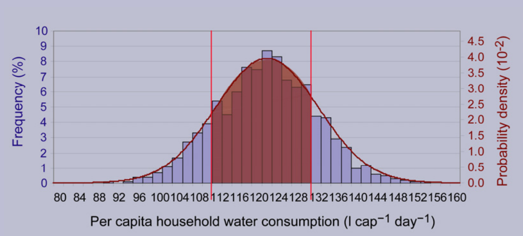
Water consumption range (l cap^{-1} day^{-1})	Data points in each range (%)
≤88	0
88-90	0.1
90-92	0
92-94	0.1
94-96	0.4
96-98	0.4
98-100	0.7
100-102	1.1
102-104	1.7
...	
152-154	0.1
154-156	0
>156	0

The sum of all bar heights corresponds to 100%. The proportion of values lying in a specific range can be estimated by adding the heights of the bars in this range based on the frequency histogram. The proportion of values between 110 and 130 l cap^{-1} day^{-1}, for example, amounts to 68%.



Water consumption in Hanoi represented as frequency histogram.

A smooth curve can be plotted through the bars of the frequency histogram: this is a **probability density function**. The total area below this curve is one. The area under the curve above a certain range corresponds to the probability of lying in this range. The shaded area in the figure below between 110 and 130 l cap^{-1} day^{-1} amounts to 0.68.



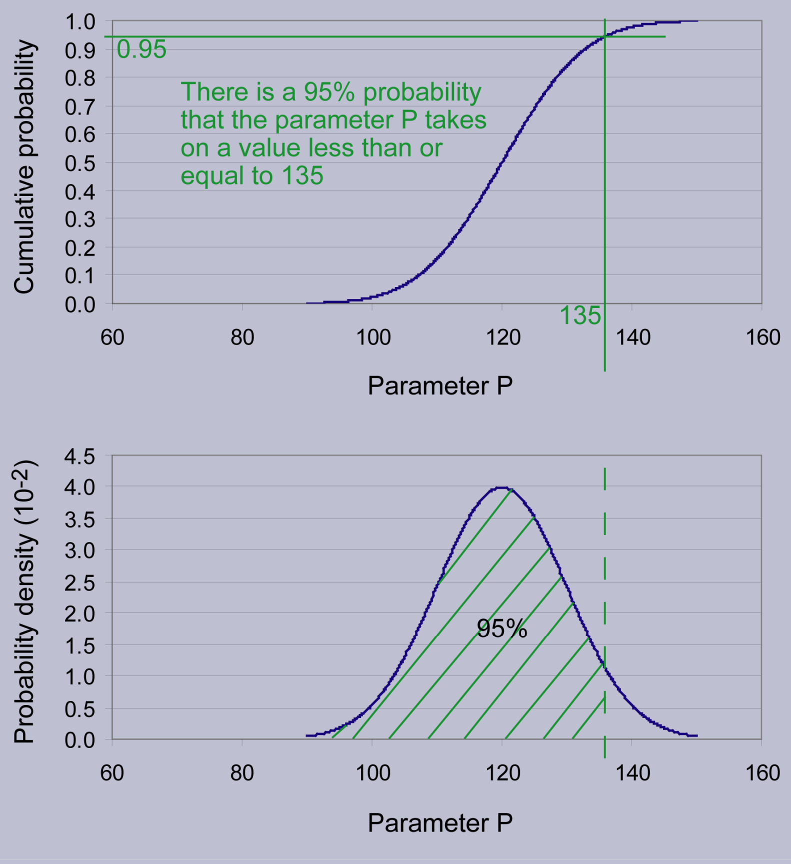
Water consumption in Hanoi represented as probability density function.

However, it is very likely that we do not have enough data points available to build a frequency histogram or a probability density function. In this case, we can use the subjectivist view to generate the data points. Assuming that household water consumption is likely to range around the mean value with fewer values much smaller or much larger that the mean, it can be described by a normal distribution. In the case of Hanoi, a mean value of 120 l cap^{-1} day^{-1} was determined on the basis of available data and a standard deviation of 10 was assumed also based on the range of available data and on the reliability of the data sources. The characteristics of the distribution function can be used to generate random data points (for example using excel) and subsequently to plot a frequency histogram or probability density function. If data is too scarce to estimate the characteristics of the probability distribution such as mean and standard deviation, assessing expert judgement is a promising method (Box 5).

Another way of expressing probabilities is the **cumulative distribution function**. It represents the probability that an uncertain quantity is less than or equal to a specific value (see figure below). The cumulative distribution function is the integral of the probability density function.

The *Palisade's DecisionTools* software (Clemen and Reilly, 2001) can help plotting probability density or cumulative probability functions.

Sources: Clemen and Reilly (2001), Morgan and Henrion (1991)



Cumulative distribution function and probability density function of the same parameter

Box 5 Expert judgment elicitation

Box 5.1 How to assess expert judgment?

If data is scarce, assessing expert judgement is a promising method to estimate model parameter values. Expert knowledge (subjective judgements) can be translated into probability distributions, i.e. *prior* probability distributions (Morgan and Henrion, 1990).

This approach usually requires the development of a protocol for expert assessment. The main steps of the approach (Morgan and Henrion, 1990; Clemen and Reilly, 2001) are described here and are followed by an example:

1. Background

The first step consists in identifying the parameters for which expert assessment is needed. A search of relevant scientific literature should be conducted to determine the extent of available scientific knowledge.

2. Identification and recruitment of experts

There is no rule related to the appropriate number of experts. However, enough experts are required to illustrate the main views (Morgan and Henrion, 1990).

3. Motivating experts

Scientists often prefer to rely on measurements rather than on subjective judgement. Since they may hesitate to express their opinions, it is important to gain the experts' confidence and raise their enthusiasm for the project.

4. Structuring and decomposition

This step explores the experts' understanding of causal relationships among the relevant parameters. The objective is to develop a general model such as an influence diagram that reflects the experts' thinking about the relationships among the parameters.

5. Probability assessment training

It is important to explain the principles of expert assessment, to provide information on the inherent biases in the process and ways to counteract those biases (see below), and to give the experts an opportunity to practice probability assessments.

6. Probability elicitation and verification

This step produces the required probability assessments and a documentation of the reasoning behind the assessments. Each expert should be asked to review the plotted distributions and provide clarifications (see example below).

7. Aggregation of experts' probability distributions

If experts' views on the same parameter differ, it is important to try to understand why they reach different conclusions. If the range of opinions has no significant consequence on the model outcome, their conclusions can be combined to obtain an average view. On the other hand, if the range of experts' opinions has a major impact on model outcome, their opinions should in most cases not be combined (Morgan and Henrion, 1990).

Additional knowledge subsequently gained through measurements for example can be combined with expert knowledge. Bayes' rule provides the mechanism for using data to update a prior probability distribution in order to obtain a posterior probability distribution (Morgan and Henrion, 1990; Clemen and Reilly, 2001).

Box 5.2 How to avoid biases?

Heuristics are simple rules, like rules of thumb we use when making judgments in the presence of uncertainty. However, they can lead to biases (Morgan and Henrion, 1990). These biases are described here as well as measures to counteract them.

Anchoring and adjustment. According to this heuristic, we tend to use the most readily available piece of information as a reference point (the "anchor") and make adjustments to it to reach our estimate (Spetzler and Staël von Holstein, 1975). A possibility to counter anchoring bias is to ask experts for extreme judgments before obtaining their likely ones (Meyer and Booker, 1991).

Availability “refers to the ease with which relevant information is recalled or visualized”. Information that made a strong impression on us or recent information is likely to be easier to recall and therefore to be given too much weight (Spetzler and Staël von Holstein, 1975). “In general, group discussion will cause the expert to think of more than just the first readily accessible information” (Meyer and Booker, 1991).

Representativeness. We tend to make probability judgments on the basis of similarity. For example we judge the probability that someone belongs to a specific group by comparing information about that person with the stereotypical member of that group (Clemen and Reilly, 2001).

Underestimating uncertainty. This “may be reduced by asking the expert to further disaggregate the parts of the question and give estimates of the quantities of interest for each small part” (Meyer and Booker, 1991).

Consequently, expert interviews must be very carefully conducted in order to avoid biases. In particular, it is recommended to disaggregate the question in different parts and then to estimate the parameters for each small part, to start by estimating extreme values before assessing mean values and to conduct group discussion to avoid basing our estimation on events that are easy to recall.

Box 5.3 Assessing nutrient transfer coefficients in septic tanks by eliciting expert judgment

The expert elicitation technique was applied in the Hanoi case study to determine prior probability distributions for nitrogen and phosphorus transfer coefficients in septic tanks – one of the most common on-site sanitation systems in developing countries.

Transfer coefficients, commonly used when modelling material flows, describe the partitioning of a substance in a process and provide the fraction of the total input of a substance transferred into a specific output good (Baccini and Bader, 1996). Transfer coefficient values, which are not necessarily constant, are substance and process-specific. Transfer coefficients are therefore defined for specific conditions (e.g. operational procedure, geographical location). The transfer coefficient for a substance i (e.g. phosphorus) through a process (e.g. septic tank) into an output good j (e.g. faecal sludge) is defined by the following equation:

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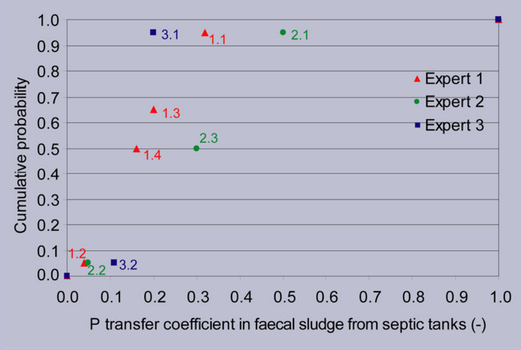
where $k^{(j)}_{i,g}$ stands for the transfer coefficient of substance i in output good g for process j, $A^{(j)}_{i,g}$ is the flow of substance i in good g generated in process j, and $\sum A_{i,r\text{-}j}$ the total input flow of substance i into process j.

Since septic tanks were designed to reduce solids and organic matter loads in the wastewater and not to remove nutrients, it is not surprising that information on nutrient removal in septic tanks is scarce.

Experts were selected on the basis of their theoretical knowledge on wastewater treatment techniques and research experience related to septic tanks. Three experts participated in the expert elicitation study on nitrogen and phosphorus partitioning, and an additional one supported the study with his knowledge on nitrogen behaviour.

Experts were first introduced to the project and to the principles of subjective probability assessment. They examined the results of a literature review on septic tanks and were given the opportunity to correct or add information. They were then asked to generally describe the qualitative mechanisms occurring in a septic tank and, in particular, the ones influencing nutrient separation in faecal sludge, effluent and gas.

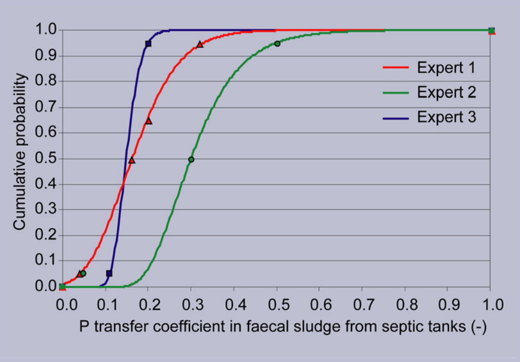
In a second step, points on the cumulative distribution function were assessed by assigning a transfer coefficient value to a given cumulative probability. To avoid anchoring, extreme points, i.e. transfer coefficient values corresponding to cumulative probabilities of 95% and 5% were first assessed. After having assessed the largest and smallest values, intermediate cumulative probabilities and their corresponding transfer coefficient values were determined. Experts were asked to explain the rationale for their assessment.



First step of the probability elicitation. Phosphorus transfer coefficient values are assigned to given cumulative probabilities starting by extreme values. Points 1.1-3.2 designate the points assessed by the experts in their order of assessment.

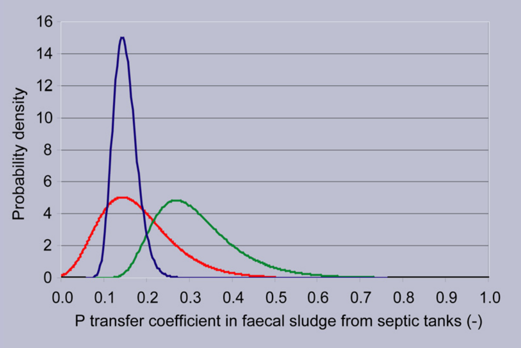
The cumulative distribution functions were then fitted through the points (non-linear regression) using the *Palisade's DecisionTools* software

(Clemen and Reilly, 2001) and assuming that transfer coefficients follow a lognormal distribution (non-negative, positively skewed, appropriate to represent large uncertainties).



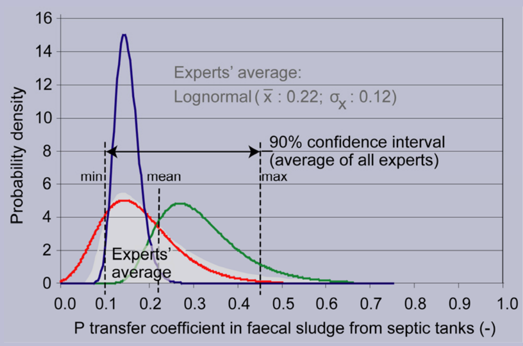
Second step of the probability elicitation. Cumulative distribution functions are subsequently fitted through the points.

The cumulative distribution functions were then converted to probability density functions (derivation of the cumulative distribution functions). Experts were invited to review the results of their interviews.



Third step of the probability elicitation. The cumulative distribution functions are converted to probability density functions (derivation of the cumulative distribution functions).

The resulting probability density functions were averaged by attaching the same weight to the results obtained by each expert.



Fourth step of the probability elicitation. Averaging of the obtained probability density functions (grey area).

The mathematical properties of the obtained theoretical distribution were subsequently used in the entire Hanoi nutrient flow model.

Source: Montangero and Belevi (2007)

Box 6 Uncertainty propagation

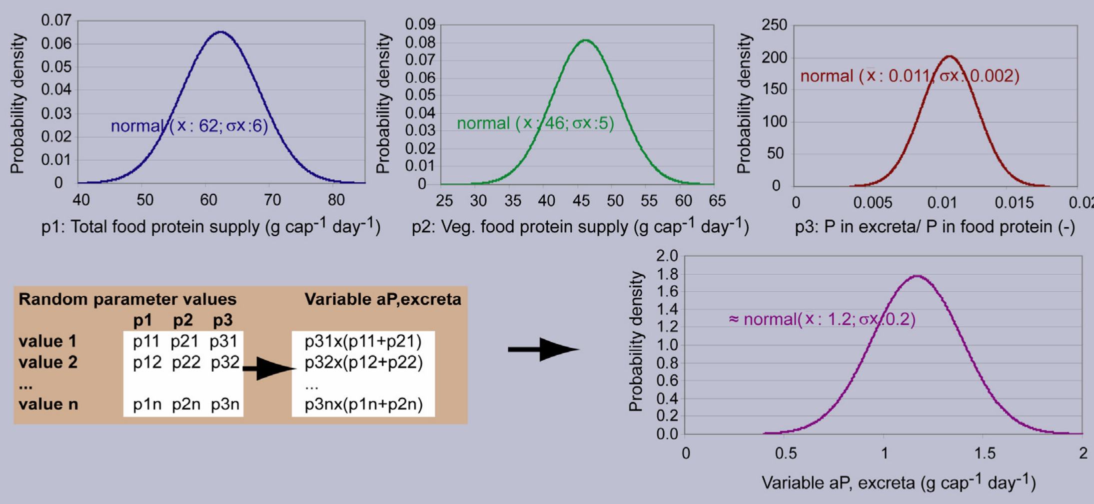
The question is how to propagate the parameter uncertainties through the model in order to assess uncertainty in model outcomes, i.e. in variables such as material flows. Many methods can be used to assess propagation of uncertainties including those under the general descriptions of analytical, approximation and numerical methods. If parameter uncertainties are large and deviate from normal distribution, it may be difficult or impossible to combine uncertainties using the conventional statistical rules (IPCC, 2000). However, the numerical **Monte Carlo method** can deal with this situation. The principle consists in generating model parameter values randomly within the parameter uncertainty distribution specified initially (i.e. more values are selected from the high probability range). Variable values are subsequently calculated based on the generated parameter values. An uncertainty distribution for the variable can be determined based on the calculated variable values. The method is computationally time-intensive, but well suited to deal with the problem of propagating and aggregating uncertainties in an extensive system (IPCC, 2000).

The following example illustrates how the Monte Carlo simulation can be used to estimate the uncertainty of the phosphorus load in human excreta on the basis of the parameter uncertainties:

As aforementioned, the phosphorus load in excreta can be expressed as a function of the total food protein (a_{TFP}) and vegetable food protein (a_{VFP}) supplied to the "household" in g cap^{-1} day^{-1} as well as of the ratio between P load in excreta and the sum of total and vegetable protein supply ($r_{P\_excreta}$):

$$a_{P,excreta} = r_{P\_excreta} \cdot (a_{TFP} + a_{VFP})$$

n random values for these three parameters can be generated, for example using excel, based on their type of probability distributions and characteristics such as mean and standard deviation. A series of n phosphorus load values can subsequently be calculated using the equation above and the n parameter values. The n phosphorus load values are classified in different intervals based on which a frequency histogram or a probability density function can be plotted (Box 4). Mean and standard deviation of the phosphorus load can now be determined.



Principles of the Monte Carlo simulation illustrated by the assessment of the phosphorus load in human excreta.

3.2.3 Calculating variable values

Material flows can now be calculated based on the defined equations and parameter values

Once parameter values are assessed, variables, i.e. stock change rates in each process and flows can be calculated using the defined equations (Chapter 3.2.1). Variable uncertainty was assessed by Monte Carlo simulation (Box 6), whereby a set of values was calculated for each variable by randomly selecting parameter values based on their probability distributions and by applying the model equation. As mentioned in Chapter 3.2.2, parameter values are determined in an iterative way: they are reassessed until all plausibility criteria are met. Variable values are thus also recalculated after each parameter reassessment.

Material flow uncertainty is assessed using Monte Carlo simulation

There are different ways of presenting results of a material flow analysis. It is important to give the audience a feeling for flow uncertainty. Flows can be represented as arrows characterized by a specific thickness in the material flow scheme, frequency histograms or probability distributions (Fig. 9). However, if the target reader does not feel comfortable with probability distributions, it may be better to represent flows as intervals using for example the 90% or 95% confidence interval (see Fig. 14). When using power point presentations, it is appropriate to represent the outcomes of different scenarios as a series of material flow schemes where the relative importance of the flows is represented by arrows of different thicknesses.

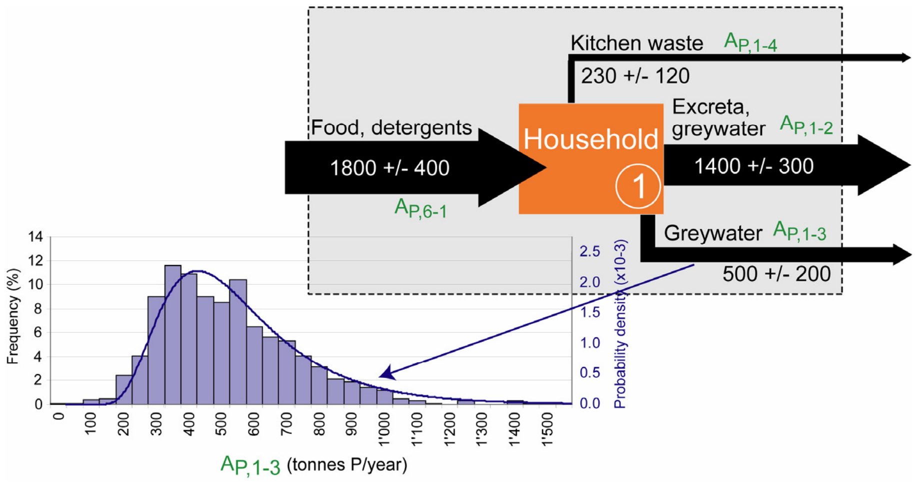


Figure 9 Phosphorus flows to and from the household process in Hanoi province. Flows are represented as arrows of different thicknesses. Means and standard deviations are indicated (tonnes P per year). One of the flows, P in greywater is also represented as frequency histogram and probability density function.

3.3 Identifying sensitive parameters

Sensitivity analysis provides insight into the most determining parameters. This helps design effective measures and select parameters requiring a more precise assessment in order to reduce variable uncertainty. This is a very important step where only limited data collection resources are available, as it reduces the number of parameters requiring further quantification.

The sensitive parameters, i.e. the most determining factors, are identified. This helps understanding the system, designing effective measures and optimizing data collection or monitoring plans

Sensitivity was determined as follows: A variable value was calculated on the basis of the parameter values whereby one of the parameters was changed by 10%, for example, and all other parameters were left unaltered. The difference between the variable value and the value obtained by changing one parameter was then determined. The procedure was repeated for all parameters influencing the given variable. This is illustrated here based on two examples: groundwater consumption and phosphorus discharge into surface water.

Groundwater consumption can be expressed as a function of the following parameters: number of inhabitants (n), per capita household water consumption (a_{HH_water}), market water consumption, industrial water consumption ($A_{5\text{-}0}$), and ratio of water lost through leakage ($r_{leakage}$). As data on market water consumption was not available, it was assessed on the basis of market wastewater generation rate (a_{MWW}), evaporation (a_{evap}), and market area (s_M):

$$A_{16-5} = (n \cdot a_{HH\_water} \cdot 365 \cdot 10^{-3} + (a_{MWW} \cdot 365 - a_{evap}) \cdot s_M \cdot 10^{-3} + A_{5-0})/(1 - r_{leakage})$$

To identify sensitive parameters, the groundwater abstraction variable was calculated by increasing one of the parameters by 10% while leaving the others unaltered and repeating this procedure for each parameter. Fig. 10 illustrates the results of the sensitivity analysis for the groundwater abstraction variable. It illustrates changes in groundwater abstraction for a 10% parameter increase. The number of inhabitants, per capita water consumption and proportion of water lost through leakage are the most sensitive parameters. Measures to reduce per capita water consumption such as water saving house installations or reuse of rainwater or greywater for toilet flushing would therefore lead to a significant reduction in groundwater withdrawal.

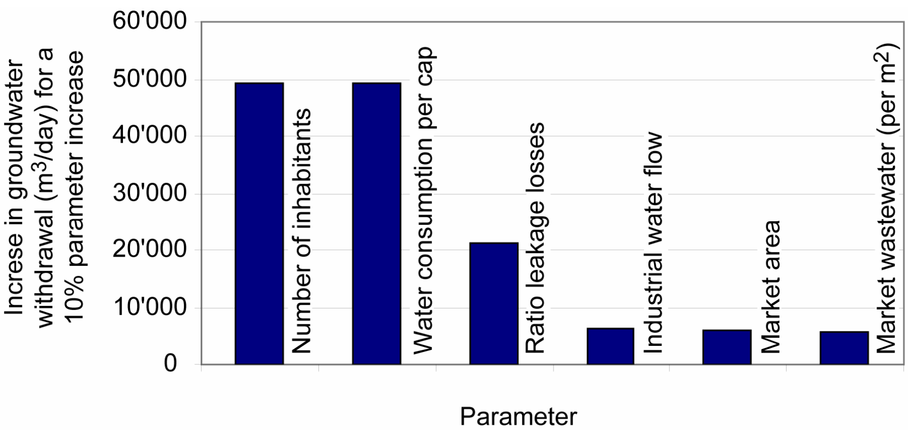


Figure 10 Effect of a 10% parameter increase on water consumption (sensitivity analysis).

The following parameters are of key influence on phosphorus flows into surface water: the number of inhabitants, ratio of septic tank effluent discharged into the drainage network, food protein amount supplied to the households, ratio between P load in excreta and food protein supply, ratio of inhabitants equipped with septic tanks, and P load in greywater (Fig. 11). Assuming that the number of inhabitants and per capita protein supply are given conditions and that the ratio between P load in excreta and food protein is a constant value, the percentage of inhabitants equipped with septic tanks as well as per capita phosphorus load in greywater appear to be key control points to regulate the phosphorus load discharged into surface water (Chapter 3.5.2).

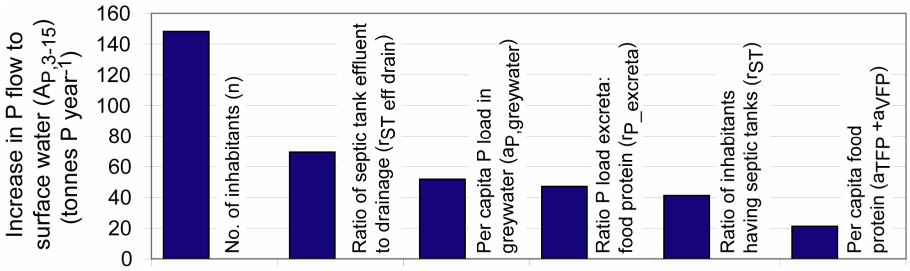


Figure 11 Effect of a 10% parameter increase on phosphorus discharge into surface water (sensitivity analysis). Only the most sensitive parameters are represented here.

Moreover, those are also the parameters on which to focus when reassessing parameter values (Chapter 3.2.2). As a whole, around 150 parameters determine phosphorus discharge. Focusing only on the most 5 or 10 sensitive ones thus considerably reduce the means required for data collection.

3.4 Assessing plausibility of parameter and variable values

3.4.1 Formulating plausibility criteria

Plausibility of model input parameters and calculated model variables is verified by crosschecking with reliable literature data or field measurements or by using overdetermined sets of equations

When dealing with scarce data, it is important to determine plausibility of model input parameters and of model outcomes by developing a set of plausibility criteria representing parameters, variables (flows and stock change rates) or relationships between parameters or variables (e.g. N/P ratios). Plausibility criteria ranges are then derived from reliable sources. Should the estimated parameter values or calculated model outcomes lie outside the plausible ranges, the assessed parameter values have to be revised.

Plausibility criteria can be based on **crosschecking**. For example, a flow value calculated using the model could be compared with the results of field measurements or literature data considered as reliable. This is illustrated by the following examples:

A wastewater generation rate of approximately 460 000 m^3 day^{-1} was estimated by Hanoi DOSTE (2003). The sum of the different wastewater flows determined by the model, i.e. industrial, household (greywater and effluent from on-site sanitation systems) and market wastewater should thus amount to:

$$A_{0-3}+A_{1-3}+A_{2-3}+A_{6-3}=460\,000\pm 15\%\ m^3\ d^{-1} \qquad \text{(Criteria 1)}$$

(assuming a 15% uncertainty interval for the DOSTE value)

Moreover, the literature provides various per capita P load ranges in excreta for different regions. In Vietnam, a P load in excreta of 0.7-1.5 g cap^{-1} day^{-1} was assessed based on information from Polprasert et al. (1981) and Heinss et al. (1998). The P flow in excreta calculated by the model on the basis of the total and vegetable food protein contained in the food supplied to the households is compared to the aforementioned literature range:

$$a_{P,excreta}=r_{P\_excreta}\cdot(a_{TFP}+a_{VFP})=0.7-1.5\ \mathrm{g\ cap^{-1}\ day^{-1}} \qquad \text{(Criteria 2)}$$

A reliable P concentration range in greywater was determined based on a field study conducted in Hanoi (Büsser, 2006). The P concentration was first approximated by literature review and compared to this range:

$$C_{P,greywater}=a_{P,greywater}/a_{greywater}\cdot 10^3=2-6\ mg\,\mathrm{l^{-1}} \qquad \text{(Criteria 3)}$$

Another possibility to establish plausibility criteria is to use an **overdetermined set of equations**. For example, we could add an equation to the ones characterizing phosphorus flows in the household process (Chapter 3.2.1): there should be no decrease or increase of the phosphorus stock in this process and thus, based on the household balance equation:

$$\frac{dM_P^{(1)}}{dt}=A_{P,5-1}+A_{P,6-1}-A_{P,1-2}-A_{P,1-3}-A_{P,1-4}-A_{P,1-17}=0\pm 15\% \qquad \text{(Criteria 4)}$$

3.4.2 Verifying the plausibility of parameter and variable values

This step compares parameter values and model outcomes with the reliable ranges determined for the plausibility criteria. If plausibility criteria are not met, parameter assessment should be repeated. Fig. 12 illustrates plausibility assessment on the basis of selected criteria. Parameters and model outcomes are represented as frequency histograms and were obtained from the first rough parameter approximation. It is assumed here that the criteria are fulfilled if at least 68% of the estimated parameter or calculated variable values lie within the plausible range. The limit of 68% was selected as it corresponds to the proportion of values ranging within one standard deviation from the mean value for a normally distributed variable (Box 3). Since the criterion for P concentration in greywater was not met (Fig. 12), it was reassessed more accurately (Chapter 3.2.2). Similarly, all sensitive parameters determined in Chapter 3.3 were reassessed more accurately until all plausibility criteria were met.

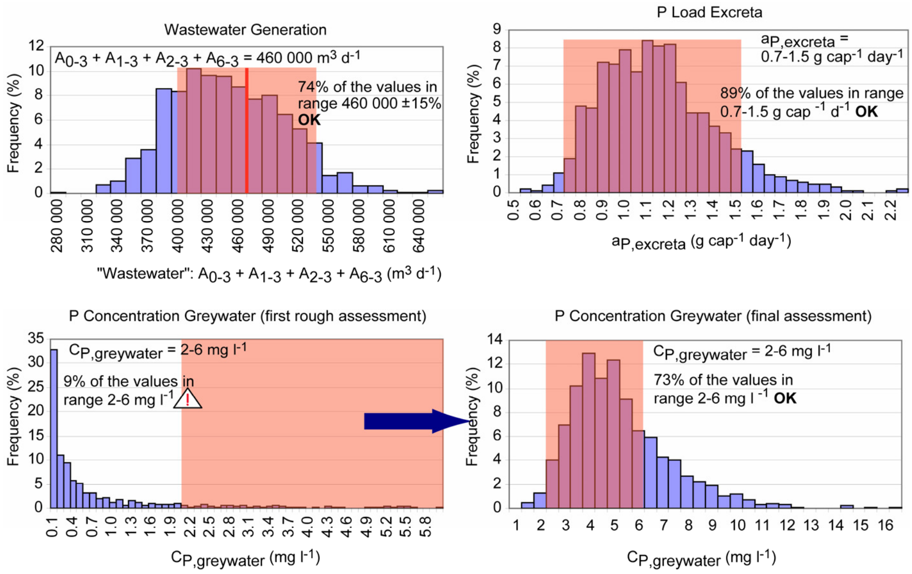


Figure 12 Plausibility assessment results obtained from selected criteria. Model outcomes are illustrated as frequency histograms and plausible ranges as rectangles. Model outcomes are based on the first parameter approximation values. Plausibility criteria and percentage of the values within the plausible ranges are indicated.

The model can be used as a basis for problem identification and scenario simulation once model parameter values and model outcomes are plausible.

3.5 Simulating the impact of measures

To simulate the impact of changes in the system, parameter values are adapted so as to describe these changes. For example, the impact of population growth is simulated by replacing the initial characteristics of the parameter "number of inhabitants" (n) with the new ones, i.e. at a specific point of time (in 10 or 20 years time, for example). Or if a new process is introduced for example, an additional model component is integrated into the existing model. A few examples are given here to illustrate the procedure.

3.5.1 Reducing water consumption

The model results reveal that the current groundwater abstraction in Hanoi province amounts to 620,000 ± 90,000 m^3 day^{-1} (Montangero et al., in press). Even though uncertainty is rather high, the model outcome compares well with the plausible range of 600,000 - 650,000 m^3 day^{-1} (Hanoi CERWASS, 2004). 60% are used for domestic consumption and a relatively large amount is lost through leakage (Fig. 13). The groundwater withdrawal rate is in the same order of magnitude as the estimated aquifer recharge rate of about 700,000 m^3 day^{-1} (Nga, 2005). Moreover, abstraction rate is steadily increasing as a result of population growth, increase in per capita water consumption and industrialisation.

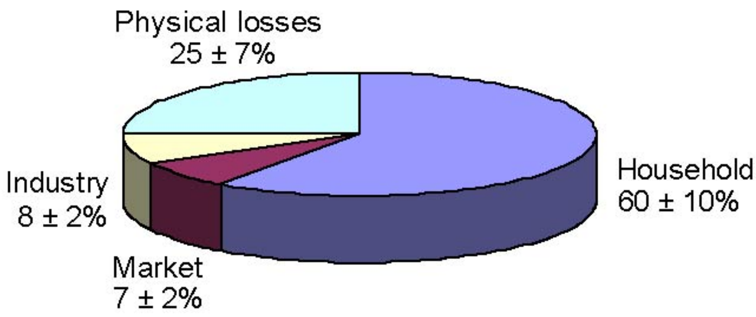


Figure 13 Groundwater consumers and their relative importance (mean ± standard deviation)

Sensitivity analysis for the groundwater abstraction variable revealed that the parameters with the greatest impact on groundwater abstraction comprise number of inhabitants, per capita water consumption and ratio of water lost through leakage (Fig. 10). Thus, assuming that population continues to grow at a high rate, measures leading to a decrease in per capita water consumption and to an improvement of the water distribution network will have the most significant influence on the amount of water withdrawn.

The model was used to determine the impact of different measures on groundwater withdrawal (see Table 3, Annex A for parameter values). In the first scenario, groundwater abstraction was simulated for the year 2015 based on unchanged current trends. Assuming that Hanoi's population continues to increase at a high rate and reaches 5 million by 2015, water consumption rises from 120 to 140 l cap^{-1} day^{-1}, market area increases by 10%, and industrial water consumption doubles, water abstraction by 2015 would almost double to 1,134,000 ± 171,000 m^3 day^{-1}. Since this figure is far above the aquifer recharge rate of 700,000 m^3 day^{-1}, urgent measures are required to guarantee adequate water supply and avoid land subsidence problems in future.

In the second scenario, the impact of a series of measures aiming at reducing water consumption was simulated for the year 2015. The results of the sensitivity analysis give first indications of the kind of measures likely to be effective. The first simulated measure consists in reducing domestic water consumption by reusing a fraction of the greywater for toilet flushing. This measure would lead to a decrease in mean water consumption from 140 to 112 l cap^{-1} day^{-1} and correspond to a 16% water abstraction decrease. The second measure consists in improving the water distribution network and thus reducing leakage from 25 to 10% and total water consumption by 17%. The third measure consists in reducing the industrial water consumption by using more water-efficient production techniques. A 30% reduction in industrial water consumption would lead to a 4% reduction in total groundwater withdrawal. By implementing all three measures, a 33% reduction in water consumption could be achieved, corresponding to 762,000 ± 106,000 m^3 day^{-1}. Interventions such as information and awareness raising campaigns, as well as introduction of financial mechanisms (incentives and/or sanctions) are preconditions for successful implementation of the aforementioned measures. However, even though groundwater abstraction rate could be reduced by a third, it would still be in the same order of magnitude as the maximal recommended withdrawal rate (Fig. 14).

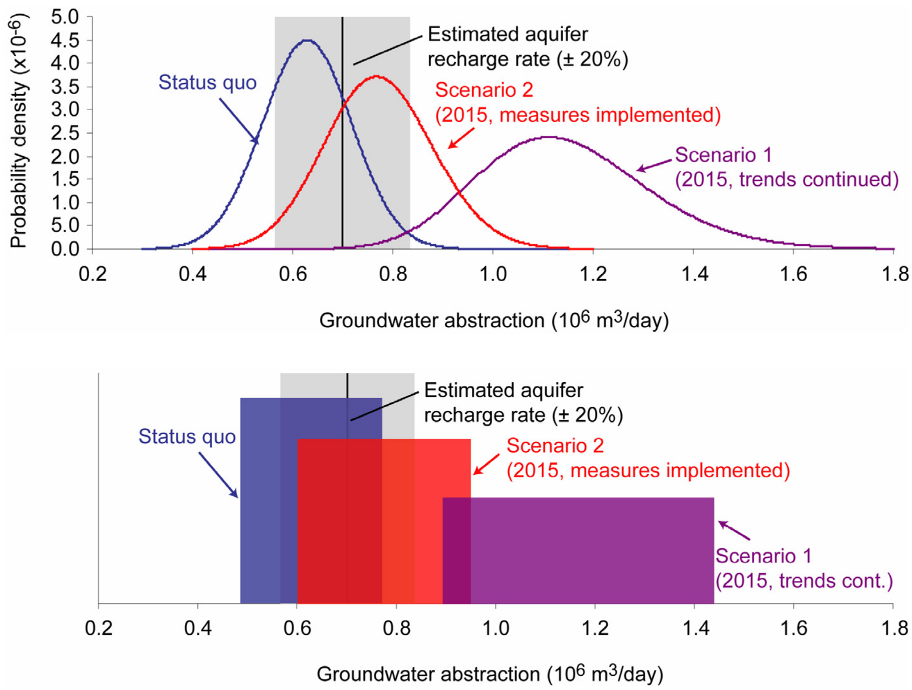


Figure 14 Groundwater abstraction today, in 2015 if current trends remain unchanged (scenario 1), and in 2015 if leakage is reduced, greywater reused for toilet flushing and industrial water consumption reduced (scenario 2). The estimated aquifer recharge rate is also illustrated assuming an uncertainty interval of ± 20%. Representation as probability density functions (top) and as ranges (90%confidence interval, bottom).

Working with probability distributions rather than mean values allows a more differentiated interpretation of the model outcomes. Knowledge of the probability distribution of groundwater abstraction in scenario 2, for example, reveals that the probability that groundwater consumption lies in the range of the aquifer recharge rate amounts to 75%. There is a 23% probability that it exceeds the aquifer recharge rate and only a 2% probability that it lies below. In order to determine whether the mean values of model outcomes representing the same variable simulated for two different scenarios differ significantly from each other, statistical tests can be performed (see Gränicher, 1996 and Gellert et al., 1966).

Use of river water as additional water supply source is currently being discussed in Hanoi. However, the fact that surface water is seriously polluted renders its treatment unaffordable. Recently, water is being piped to Hanoi from the 70-km distant Da River. Imports of less polluted surface water from other regions are likely to increase in the near future. Yet this could lead to conflicts among consumers and increase water production costs. Surface water quality improvements in and around Hanoi could render its use as water supply source more affordable. Furthermore, promoting the development of neighbouring small to medium-size cities, as proposed in the Master Plan for Development of Hanoi (Hau, 2004), could limit groundwater consumption increase in Hanoi. Groundwater replenishment measures should also be discussed.

3.5.2 Reducing phosphorus discharge into surface water

Double-pit latrine with urine diversion (Viet Tri, Vietnam): latrine floor with openings to the two chambers for faeces, a urine diversion channel and a urine pot.

Besides reducing the amount of phosphorus-containing detergent used in the households, replacing septic tanks by urine diversion latrines seems promising as urine diversion latrines, unlike septic tanks, can immobilise most of the nutrients either as dehydrated faeces or urine; both reusable as fertiliser in peri-urban agriculture (Montangero and Belevi, 2007). The model allows to simulate the impact on nutrient load discharged into surface water when septic tanks are replaced by urine diversion latrines. Fig. 15 illustrates P flows today and in the year 2015 assuming trends remain unchanged, as well as for the year 2015 assuming the inhabitants equipped with septic tanks use urine diversion latrines.

Despite the high uncertainty, a comparison of the probability density functions reveals a significant reduction between the nutrient flows simulated for the year 2015 and for the same year based on the assumption that Hanoi's inhabitants use urine diversion latrines. The phosphorus flow simulated for the year 2015 amount to 2,700 ± 600 tonnes year^{-1}. Replacing septic tanks by urine diversion latrines would reduce it to 1,500 ± 400 tonnes year^{-1}, corresponding to a 45 ± 11% reduction. The main parameters determining this flow are contained in Table 4 (Annex A). Results can be illustrated for example as probability distributions (Fig. 15) or as arrows of different thicknesses in material flow schemes (Fig. 16).

Ecosan latrine in Ha Tay Province, Vietnam. © CEETIA.

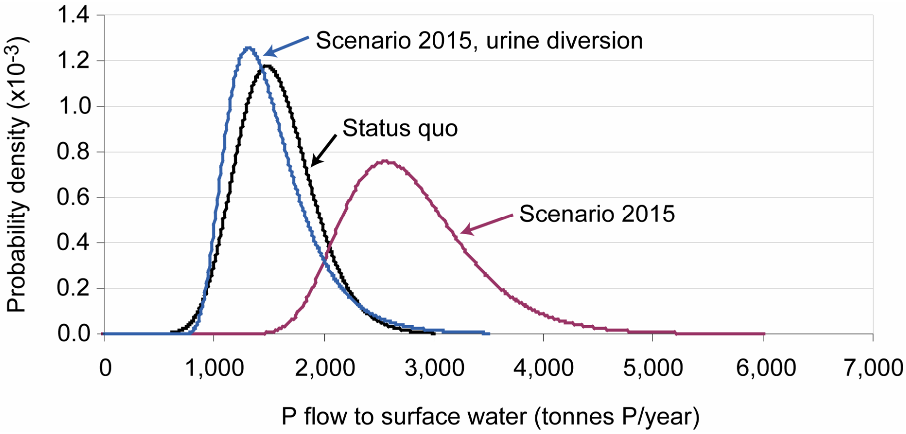


Figure 15 Simulated phosphorus flows to surface water today, in 2015 assuming unchanged trends and in 2015 assuming septic tanks are replaced by urine diversion latrines.

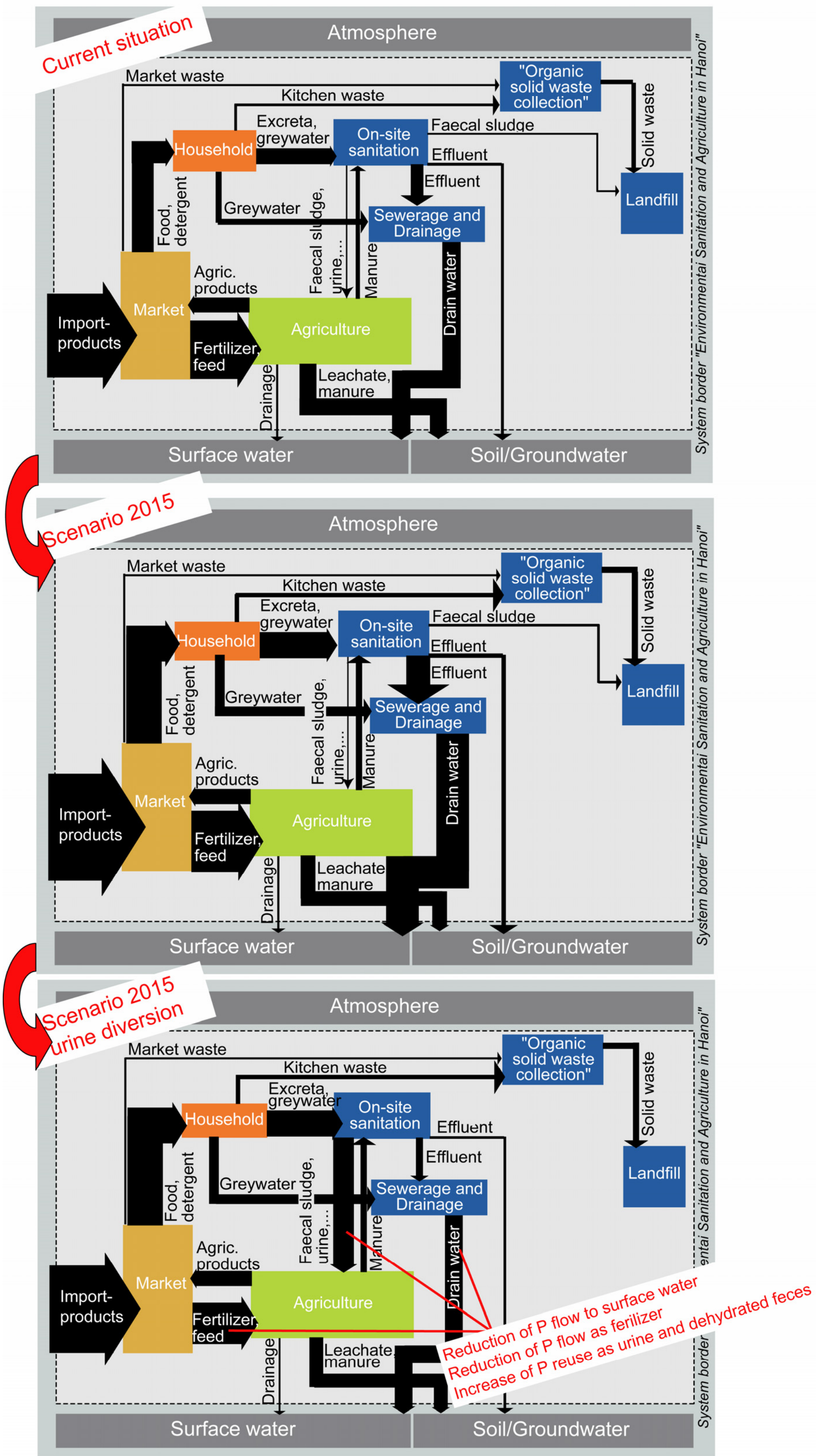


Figure 16 Relative importance of the main phosphorus flows in Hanoi province today (top), in 2015 assuming unchanged current trends (center), and in 2015 assuming that septic tanks are replaced by urine diversion latrines (bottom).

3.5.3 Reducing artificial phosphorus fertilizer use

The calibrated model can also be used to simulate the effect of different measures on phosphorus recovery in Hanoi's peri-urban agriculture (Montangero et al., in press).

In a first step, it is useful to have an overview of the distribution of the phosphorus load in the different waste products (Fig. 17). Phosphorus quantity in waste products (generated in households, markets, industries, and peri-urban agriculture) amount to 4,400 ± 800 tonnes P year^{-1}. It is important to note that a considerable amount of the load is contained in agricultural waste products: 38% in manure and 6% in crop residues. Furthermore, 36 ± 13% of the phosphorus load are contained in liquid waste products (on-site sanitation effluent and greywater).

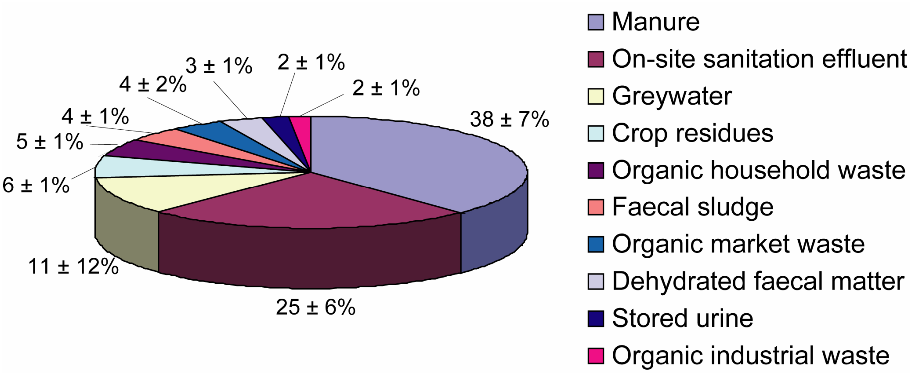


Figure 17 Fractions of the phosphorus load in waste products.

Septic tank effluent in Hanoi is one of the main contributors of the nutrient load in waste products: A high percentage of the nutrient entering the septic tank with urine and faeces leave the tank with the liquid effluent (Montangero and Belevi, 2007). Moreover, septic tank effluent in Hanoi is mainly discharged into the sewerage and drainage network instead of infiltrating into the ground through leaching pits or fields.

Currently, 1,000 ± 200 tonnes P year^{-1} or 23 ± 5% of the phosphorus load in waste products are recovered as organic fertiliser, irrigation water, livestock or fish feed through reuse of waste products. The remaining waste products are landfilled or discharged into the drainage network or on open grounds, leading not only to a loss of valuable resources but also to environmental pollution (Fig. 18). The phosphorus recovered corresponds to 18 ± 3% of the total phosphorus actually used for food production in Hanoi province. The residual phosphorus demand is covered by artificial fertiliser and commercial livestock feed (Fig. 19).

Food production within the urban and peri-urban area of the province covers about 44% of the food demand (Anh et al., 2004). Only 11 ± 3% of the P amount required to produce food for the entire population of Hanoi (agriculture in and outside Hanoi) are currently covered by waste products generated in the province (Fig. 19).

As aforementioned, a large fraction of the phosphorus load in waste products is contained in the effluent from on-site sanitation installations and in greywater. However, since rainwater covers most of the irrigation water requirement during the rainy season (May to October), only a small fraction of this wastewater can be recovered. A large amount of wastewater therefore ends up in the river. The question is therefore whether the environmental sanitation and agricultural system can be altered so as to increase phosphorus recovery.

The model was used to tackle this question and to analyse the impact of changes in the environmental sanitation and agricultural system on phosphorus recovery. Extreme scenarios were selected to clearly reveal the impact of policy changes. Scenario 1 describes the situation for the year 2015 and assumes unchanged current trends. Scenario 2 describes the situation for the year 2015 where Hanoi's septic tanks are assumed to be replaced by urine diversion latrines. Furthermore, an increased organic fertiliser application rate is assumed. Scenario 3 describes the situation for the year 2015 and assumes that Hanoi's population eliminates meat from its diet. Protein intake is compensated by a higher consumption of fish, vegetables, beans, soybean, and nuts. Fig. 18 illustrates the phosphorus loads in waste products recovered for food production as well as landfilled or discharged into the environment. Fig. 19 contains the phosphorus load applied in Hanoi's peri-urban agriculture (in waste products, commercial livestock feed and artificial fertiliser) as well as the phosphorus load applied outside Hanoi to produce food consumed in Hanoi. Figs. 18 and 19 compare the aforementioned phosphorus loads in the different scenarios (see Table 3 in Annex A for parameter values).

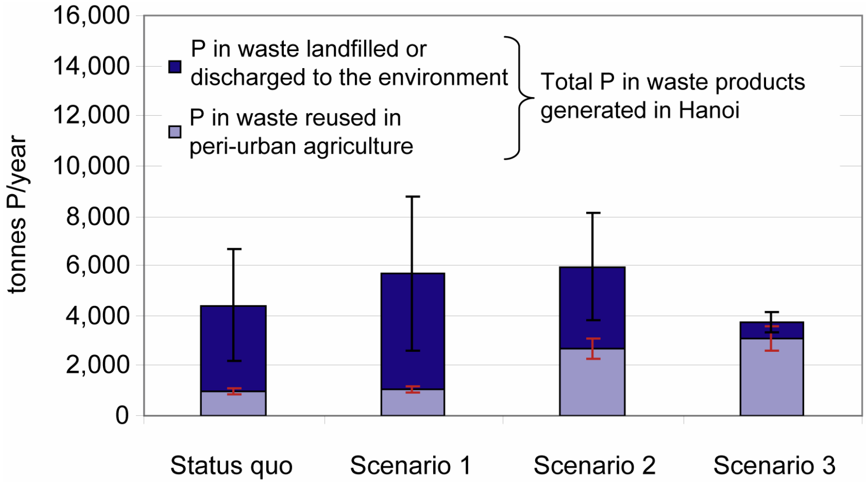


Figure 18 Phosphorus loads in waste products generated in Hanoi and reused in peri-urban agriculture as well as landfilled or discharged into the environment. The sum corresponds to the total amount of phosphorus in waste products generated by Hanoi's households, markets, industries, and agriculture. The error bars represent the standard deviation.

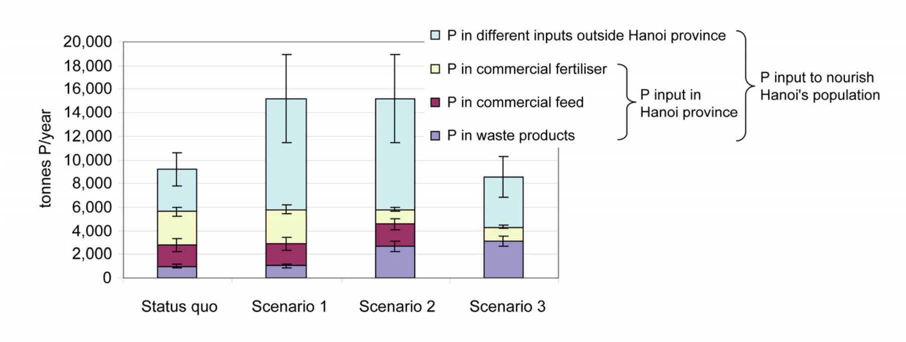


Figure 19 Phosphorus loads in waste products generated in Hanoi and reused in peri-urban agriculture, in artificial fertilizer and commercial livestock feed used in Hanoi, and in different agricultural inputs applied outside Hanoi to produce food for Hanoi's population. The error bars represent the standard deviation.

Uncertainty in Figs. 18 and 19 is represented by the standard deviation. Simulation results illustrated in these figures are discussed in the following paragraphs. Replacing septic tanks by urine diversion latrines has a considerable impact on the amount of nutrient that could potentially be recovered as organic fertiliser for food production. The amount of P that could potentially be recycled amounts to 1,000 ± 200 tonnes P year^{-1} in 2015, and 2,700 ± 500 tonnes P year^{-1} if septic tanks are replaced by urine diversion latrines. This amount would cover 17 ± 3% and 46 ± 9% of the P required as agricultural inputs, respectively. The quantity of required artificial fertiliser would correspondingly decrease 57 ± 16%.

Some 50% of the nutrients would still not be recovered and thus end up in landfill or the environment. The results reveal that modifying the agricultural production system would also have a considerable impact on nutrient recovery. A vegetarian based agricultural system would lead to a further increase in P recovery. 3,000 ± 400 tonnes P year^{-1}, corresponding to 69 ± 12% of the required phosphorus input would be recovered. Artificial P fertilizer would be reduced by 60 ± 16%.

Moreover, a considerable reduction in the total amount of nutrients required to nourish the entire population of Hanoi (peri-urban agriculture in Hanoi and neighbouring provinces) would also occur in the vegetarian scenario (Fig. 19). This is attributed to the fact that P use efficiency (required P input as fertilizer or feed per unit P produced as crop or meat) for animal production are lower than for crop production. In other words, the production of 1 kg of P as meat requires far more P input than the production of 1 kg of P as crop.

4 Summary and conclusions

When developing and calibrating material flow models with limited data availability, the following points should be taken into account:

- The model should be developed so as to minimize the number of parameters. Moreover, parameters difficult to quantify should be replaced by others more easily assessed or measured.

- To consider uncertainties, model parameters can be expressed as probability distributions. Monte Carlo simulation can subsequently be conducted to assess variables uncertainty.

- A set of plausibility criteria should be established and plausibility criteria ranges determined based on values from reliable sources.

- Application of an iterative approach is of key importance. Parameter values should first be assessed on the basis of a literature review and by eliciting expert judgement. If parameter values or calculated variables are not plausible or if the uncertainty of key variables is larger than targeted, parameter values should be reassessed more accurately. Sensitivity analysis allows to optimize data collection through identification of the parameters requiring further assessment.

- Eliciting expert judgement enhances the understanding of specific system parts and provides prior probability distributions for unknown model parameters. The prior probability distributions can be updated subsequently with data from field measurements. It is a very promising method if data availability is limited but sound expert knowledge is available.

The calibrated model developed to describe water and nutrient flows in the environmental sanitation system of Hanoi can be applied to assess the impact of different system changes, such as population growth, changes in sanitation infrastructure, consumption patterns, and reuse practices, on groundwater consumption, nutrient discharge into the environment and nutrient recovery. Sensitivity analysis allows to determine the model parameters that mostly influence the key variables. Insight into sensitive parameters thus helps propose effective measures to optimise water and nutrient management.

With regard to groundwater abstraction in Hanoi, the most sensitive parameters comprise: the number of inhabitants, per capita water consumption and ratio of water lost through leakage. By improving the water distribution network, reusing greywater for toilet flushing and increasing water efficiency of industrial processes, groundwater abstraction could be reduced by a third. However, even if these measures were implemented, groundwater abstraction would still be of the same order of magnitude as the aquifer recharge rate due to the expected high population growth. As a result of lower surface water quality, treatment is likely to be rather expensive and cleaner surface water imports from other provinces could lead to conflicts. Strategies, such as promoting the development of small to medium-size satellite cities, and thus reducing Hanoi's population growth as well as replenishing groundwater and improving surface water quality, should therefore be further discussed.

The key parameters influencing phosphorus flows into surface water include: the number of inhabitants, ratio of septic tank effluent discharged into the drainage network, ratio of inhabitants equipped with septic tanks, and per capita P load in greywater. Analysis of simulation results reveal that by replacing septic tanks with urine diversion latrines, a reduction in P to surface water of 45 ± 11% could be achieved.

Assuming that the additional amount of "organic fertiliser" (stored urine and dehydrated faeces) obtained by introducing urine diversion latrines would be used for food production in Hanoi's peri-urban area, the percentage of phosphorus demand covered by waste products in Hanoi's peri-urban agriculture would increase from 17 ± 3% to 46 ± 9%.

About 50% of the nutrient demand will still have to be covered by commercial fertilizer and livestock feed. Additional simulation results reveal that not only the type of sanitation infrastructure and reuse practices but also the agricultural system play a key role in increasing nutrient recovery. A vegetarian-based agricultural system would further increase the percentage of phosphorus demand covered by waste products (69 ± 12%). Both nutrient discharge into the environment and use of commercial fertilizer and livestock feed would consequently be reduced.

Working with probability distributions increases model information output and can hinder drawing incorrect conclusions. It enables a preliminary assessment as to whether or not simulated impacts of two different scenarios are of the same range. Should probability distributions overlap, a statistical test (for example a Student's t-test) can be performed to assess whether mean values of both scenarios differ significantly at a selected confidence interval. Use of the tool in Hanoi's case study allowed to document how careful characterisation and analysis of uncertainty allow to draw conclusions despite large uncertainties.

Application of the tool to Hanoi's case study allowed to calibrate the water and nutrient flow model and revealed potential strategies to optimize water and nutrient management despite limited data availability. The tool therefore proved to support the planning of environmental sanitation options contributing to minimize resource consumption and environmental pollution despite limited data availability.

However, by including additional processes such as further wastewater, faecal sludge and solid waste treatment options, a generic model would be achieved, which could be more easily adapted to the environmental sanitation and agricultural system of any urban region in developing and transition countries.

Moreover, consideration of further indicators would be valuable. In many regions, agricultural soils are poor in organic matter. Application of treated waste products such as compost supplies organic matter and improves soil quality. Assessment of organic matter flows could therefore also be useful to optimize resource management. However, reuse of waste products is also associated with risks. The fate of pathogenic microorganisms, hormone disrupting substances and antibiotics possibly contained in human excreta should thus be determined. Furthermore, the energy saving and energy generation potential linked to changes in the environmental sanitation and agricultural system, such as reduction of fertilizer consumption and biogas production, would be interesting to assess.

Determining the ecological sustainability indicators, such as the assimilative capacity of rivers, would contribute to assess the proposed scenarios. For example, if the simulated nutrient load discharged into the river still exceeds the assimilative capacity, further measures should be developed.

Finally, the planning of appropriate environmental sanitation systems should be carried out by the local stakeholders considering aspects such as people's perceptions, demand for improved environmental sanitation, acceptance of new options, health impact, costs, willingness and ability to pay, resource consumption and environmental pollution. It is therefore recommended to integrate this tool into a broader planning approach such as the HCES approach. However, a more differentiated MFA procedure may be required to include different spatial zones (household, neighbourhood, district, city, and region) and neighbourhood types characterized by different income levels, housing density, existing infrastructure and services, and presence of factories or farms. The next step should thus consist in testing and further adapting the tool as part of a comprehensive environmental sanitation planning project.

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Annex A

Table 1 Parameter values for the "household" phosphorus flows (first approximation!)

Symbol	Description	Unit	Uncertainty distribution (mean; standard deviation)	Reference
n	Number of inhabitants	1000 inhabitants	Normal (3,100; 400)	a
a_{food_rice}	Rice consumption	kg cap^{-1} year^{-1}	Normal (125; 10)	b and c
$a_{food\_cereals}$	Other cereals consumption	kg cap^{-1} year^{-1}	Normal (13.7; 1)	b and c
a_{food_starch}	Starchy roots consumption	kg cap^{-1} year^{-1}	Normal (5.3; 0.5)	b and c
a_{food_pulses}	Pulses consumption	kg cap^{-1} year^{-1}	Normal (29.3; 3)	b and c
a_{food_veg}	Vegetables consumption	kg cap^{-1} year^{-1}	Normal (89.8; 9)	b and c
a_{food_fruits}	Fruits consumption	kg cap^{-1} year^{-1}	Normal (45.1; 4)	b and c
$a_{food\_red\_meat}$	Pork and other red meat consumption	kg cap^{-1} year^{-1}	Normal (27.6; 3)	b and c
$a_{food\_poultry}$	Poultry meat consumption	kg cap^{-1} year^{-1}	Normal (6.5; 0.5)	b and c
$a_{food\_milk\_eggs}$	Milk and eggs consumption	kg cap^{-1} year^{-1}	Normal (9.6; 1)	b and c
a_{food_fish}	Fish, seafood consumption	kg cap^{-1} year^{-1}	Normal (15.9; 1)	b and c
a_{food_mis}	Miscellaneous food consumption	kg cap^{-1} year^{-1}	Normal (26.9; 2)	b and c
$c_{P,food\_rice}$	P content rice	g kg^{-1}	Normal (3.5; 0.3)	d
$c_{P,food\_cereals}$	Average P content other cereals	g kg^{-1}	Normal (2.5; 0.2)	d
$c_{P,food\_starch}$	Average P content starchy roots	g kg^{-1}	Normal (0.4; 0.1)	d
$c_{P,food\_pulses}$	Average P content pulses	g kg^{-1}	Normal (1.7; 0.2)	d
$c_{P,food\_veg}$	Average P content vegetables	g kg^{-1}	Normal (0.4; 0.1)	d
$c_{P,food\_fruits}$	Average P content fruits	g kg^{-1}	Normal (0.2; 0.1)	d
$c_{P,food\_red\_meat}$	P content red meat	g kg^{-1}	Normal (1.7; 0.2)	d
$c_{P,food\_poultry}$	P content poultry meat	g kg^{-1}	Normal (2; 0.2)	d
$c_{P,food\_milk}$	P content milk	g kg^{-1}	Normal (1; 0.1)	d
$c_{P,food\_eggs}$	P content eggs	g kg^{-1}	Normal (2.2; 0.2)	d
$c_{P,food\_fish}$	Average P content fish, seafood	g kg^{-1}	Normal (1.8; 0.2)	d
$c_{P,food\_mis}$	Average P content of the different food items	g kg^{-1}	Normal (1.2; 0.1)	d
a_{det}	Detergent consumption	kg cap^{-1} year^{-1}	Normal (11; 3.5)	Assumption
$c_{P,det}$	P content in detergent	g kg^{-1}	Lognormal (1.5; 0.75)	e
a_{TFP}	Total food protein in food supply	g cap^{-1} day^{-1}	Normal (62.3; 6)	c
a_{VFP}	Vegetable food protein in food supply	g cap^{-1} day^{-1}	Normal (46.2; 5)	c
$r_{P\_excreta}$	Ratio between P load in excreta and total+vegetable food protein supply	-	Normal (0.011; 0.002)	f
$a_{P,greywater}$	P load in greywater	g P cap^{-1} day^{-1}	Lognormal (0.15; 1)	g,h,i,j,k,l,m,n,o,p
r_{grey_ST}	Ratio of greywater discharged into septic tanks related to total greywater	-	Lognormal (0.1; 0.1)	Assumption
$a_{P,kitchen\ waste}$	P load in kitchen waste	g P cap^{-1} day^{-1}	Lognormal (0.2; 0.1)	g,q,r,s,t

References: [a] HSO (2002) and assuming 15% unregistered inhabitants; [b] Anh et al. (2004); [c] FAOSTAT (2004); [d] FAO (1972); [e] Gumbo and Savenije (2002); [f] Jönsson et al. (2004); [g] Schouw et al. (2002b); [h] Henze (1997); [i] Eriksson et al. (2002); [j] Lindstrom (2000); [k] Funamizu et al. (2002); [l] Ridderstolpe (2004); [m] Gajurel and Otterpohl (2002); [n] U.S. EPA (2002); [o] Czemiel (2000); [p] Gumbo (1999); [q] Diaz et al. (1996); [r] Rytz (2001); [s] Strauss et al. (2003); [t] Sinsupan (2004)

Table 2 Reassessed parameter values for the "household" phosphorus flows

Symbol	Description	Unit	Uncertainty distribution (mean; standard deviation)	Reference
a_{det}	Detergent consumption	kg cap^{-1} year^{-1}	Normal (3.4; 1.5)	a
$c_{P,det}$	P content in detergent	g kg^{-1}	Lognormal (43; 3)	a
$a_{P,greywater}$	P load in greywater	g P cap^{-1} day^{-1}	Lognormal (0.5; 0.2)	a

References: [a] Büsser (2006)

Table 3 Parameter values for groundwater consumption in the status quo, scenario 1 and scenario 2

Parameter	Unit	Uncertainty distribution (mean; standard deviation)		
		Status quo	Scenario 1	Scenario 2
Number of inhabitants	mill. inhabitants	Normal (3.1; 0.4) [a]	Normal (5; 0.6)	Normal (5; 0.6)
Household water consumption	l cap^{-1} day^{-1}	Normal (120; 10) [b]	Normal (140; 14)	Normal (112; 11)
Industrial water consumption	mill. m^3 year^{-1}	Lognormal (17; 5) [c]	Lognormal (35; 10)	Lognormal (24; 7)
Ratio of water lost through leakage	-	Normal (0.25; 0.05) [d]	Normal (0.25; 0.05)	Normal (0.10; 0.02)
Market wastewater generation rate	l m^{-2} day^{-1}	Lognormal (70; 20) [e]	Lognormal (70; 20)	Lognormal (70; 20)
Evaporation	mm year^{-1}	Normal (1262; 100) [f]	Normal (1262; 100)	Normal (1262; 100)
Market area	thousand m^2	Normal (610; 15) [g]	Normal (670; 18)	Normal (670; 18)

[a] HSO (2002) and assuming 15% unregistered inhabitants; [b] Hanoi CERWASS (2004) and Büsser (2006); [c] Osterwalder (2006); [d] WHO (2001); [e] Sinsupan (2004); [f] Müller (1983); [g] Hanoi City PC (2005)

Table 4 Parameter values for phosphorus discharge and recovery in the status quo, scenario 1, 2 and 3 (main parameters only)

Parameter	Unit	Uncertainty distribution (mean; standard deviation)			
		Status quo	Scenario 2015	Scenario 2015, urine diversion	Scenario 2015, urine diversion, no meat production
Number of inhabitants	mill. inhabitants	Normal (3.1; 0.4)[a]	Normal (5; 0.6)	Normal (5; 0.6)	Normal (5; 0.6)
Industrial wastewater flow	mill. tonnes year^{-1}	Log. (15 ; 4.5) [b]	Log. (18; 5)	Log. (18; 5)	Lognormal (18; 5)
Ratio between greywater discharged into septic tanks and total greywater	-	Log. (0.1; 0.1)	Log. (0.1; 0.1)	Log. (0.001; 0.0005)	Log. (0.001; 0.0005)
Per capita household water consumption	l cap^{-1} day^{-1}	Normal (120; 10) [c and d]	Nor. (140; 14)	Nor. (114; 11)	Normal (114; 11)
Per capita water	l cap^{-1} day^{-1}	Normal (2; 0.2)	Normal (2; 0.2)	Normal (2; 0.2)	Normal (2; 0.2)

consumption for drinking					
Per capita water consumption for toilet flushing (pour or low flush)	l cap^{-1} day^{-1}	Normal (20; 2.5) [e]	Normal (20; 2.5)	Normal (20; 2.5)	Normal (20; 2.5)
Per capita water consumption for toilet flushing (WC flush)	l cap^{-1} day^{-1}	Normal (40; 5) [e]	Normal (40; 5)	Normal (40; 5)	Normal (40; 5)
Ratio of inhabitants equipped with septic tank (WC flush)	-	Normal (0.15; 0.02) [c and f]	Normal (0.55; 0.06)	Normal (0.001; 0.0005)	Normal (0.001; 0.0005)
Ratio of inhabitants equipped with septic tank (pour flush)	-	Normal (0.48; 0.07) [c and f]	Normal (0.25; 0.05)	Normal (0.05; 0.01)	Normal (0.05; 0.01)
Ratio of inhabitants equipped with septic tanks or biogas latrines receiving pig slurry	-	Normal (0.05; 0.01) [c and f]	Normal (0.05; 0.01)	Normal (0.045; 0.01)	Normal (0.045; 0.01)
Ratio of inhabitants equipped with pour flush infiltration latrines	-	Normal (0.01; 0.002) [c and f]	Normal (0.005; 0.001	Normal (0.005; 0.001)	Normal (0.005; 0.001)
Ratio of inhabitants equipped with dry single pit latrines related to total No. of inhabitants	-	Nor. (0.13; 0.02) [c and f]	Nor. (0.1; 0.02)	Nor. (0.1; 0.05)	Nor. (0.1; 0.05)
Ratio of inhabitants equipped with dry double pit latrines with urine diversion related to total No. of inhabitants	-	Normal (0.16; 0.04) [c and f]	Nor. (0.05; 0.04)	Nor. (0.8; 0.1)	Normal (0.8; 0.1)
Ratio or inhabitants equipped with bucket latrines related to total No. of inhabitants	-	Normal (0.02; 0.01) [c and f]	Log. (0.0001; 0.0001)	Log.(0.0001; 0.0001)	Log. (0.0001; 0.0001)
Excreta generation rate	kg cap^{-1} day^{-1}	Log. (1.1; 0.2) [g]	Log.(1.1; 0.2)	Log.(1.1; 0.2)	Log. (1.1; 0.2)
Ratio between fattener and total number of pigs	-	Normal (0.9; 0.02) [h]	Normal (0.9; 0.02)	Normal (0.9; 0.02)	Normal (0.9; 0.02)
Fattener manure generation rate	kg head^{-1} cycle^{-1}	Normal (1700; 150) [i]	Normal (1700; 150)	Normal (1700; 150)	Normal (1700; 150)
Sow manure generation rate (including piglets)	kg head^{-1} year^{-1}	Normal (16 200; 1500) [i]	Normal (16 200; 1500)	Normal (16 200; 1500)	Normal (16 200; 1500)
Ratio of septic tank effluent discharged into drainage	-	Normal (0.9; 0.05)	Normal (0.9; 0.05)	Normal (0.9; 0.05)	Normal (0.9; 0.05)
Market surface area	m^2	Normal (610 000; 15 000) [j]	Nor. (670,000; 18,000)	Nor. (670,000; 18,000)	Normal (670,000; 18,000)
Market wastewater generation rate	l m^{-2} day^{-1}	Log. (70; 20) [k]	Log. (70; 20)	Log. (70; 20)	Log. (70; 20)
Cau Dien composting plant capacity	tonnes year^{-1}	Normal (13 260; 1500) [l]	Normal (16,000; 1,500)	Normal (16,000; 1,500)	Normal (160,000; 15,000)
Ratio between actual and total capacity	-	Log. (0.8; 0.1) [m]	Log. (0.8; 0.1)	Log. (0.8; 0.1)	Log. (0.9; 0.1)
Leachate production per tonne compost	tonne tonne^{-1}	Normal (0.26; 0.05) [l]	Normal (0.26; 0.05)	Normal (0.26; 0.05)	Normal (0.26; 0.05)
Ratio of leachate recycled in the composting process	-	Normal (0.4; 0.1) [l]	Nor. (0.4; 0.1)	Nor. (0.4; 0.1)	Nor. (0.4; 0.1)
Ratio between groundwater infiltration and wet drainage flow (during rainy season)	-	Log. (0.2; 0.1) [e, n, o, p]	Log. (0.2; 0.1)	Log. (0.2; 0.1)	Log. (0.2; 0.1)
Annual wet weather fraction	months months^{-1}	Normal (0.6;0.05) [q]	Normal (0.6;0.05)	Normal (0.6;0.05)	Normal (0.6;0.05)
Rain	mm year^{-1}	Normal (1682; 200) [q]	Normal (1682; 200)	Normal (1682; 200)	Normal (1682; 200)
Evaporation	mm year^{-1}	Normal (1262; 100) [q]	Normal (1262; 100)	Normal (1262; 100)	Normal (1262; 100)
Drainage catchment area	ha	Normal (22 000;	Normal (22 000;	Normal (22 000;	Normal (22 000;

		2000) [a]	2000)	2000)	2000)
Ratio impervious drainage catchment area	-	Normal (0.3; 0.1)	Nor.(0.3; 0.1)	Nor. (0.3; 0.1)	Normal (0.3; 0.1)
Ratio between drainage water exfiltration and dry weather drainage flow (during dry season)	-	Log. (0.1; 0.05) [n,e]	Log.(0.1; 0.05)	Log. (0.1; 0.05)	Log. (0.1; 0.05)
Annual dry weather fraction	months $months^{-1}$	Nor. (0.4; 0.05) [q]	Nor. (0.4; 0.05)	Nor. (0.4; 0.05)	Nor. (0.4; 0.05)
Irrigation water requirement (weighted average for the different crops)	mm	Normal (72; 10) [r]	Normal (72; 10)	Normal (72; 10)	Normal (72; 10)
Total sown area	ha	Normal (64 300; 3000) [h]	Normal (64 300; 3000)	Normal (64 300; 3000)	Normal (69 000; 3200)
Ratio agricultural drainage water recycled for irrigation	-	Log. (0.2; 0.1)	Log. (0.2; 0.1)	Log. (0.2; 0.1)	Log. (0.2; 0.1)
Ratio urban drainage water in irrigation water	-	Log. (0.07; 0.03) [s]	Log.(0.07; 0.03)	Log.(0.07; 0.03)	Log. (0.07; 0.03)
P flow in industrial wastewater	kg P $year^{-1}$	Lognormal (50 000; 15 000) [b]	Log. (60,000; 18,000)	Log. (60,000; 18,000)	Log.(60,000; 18,000)
P load in greywater	g P cap^{-1} day^{-1}	Lognormal (0.5; 0.2) [d]	Lognormal (0.5; 0.2)	Lognormal (0.5; 0.2)	Lognormal (0.5; 0.2)
P transfer coefficient in faecal sludge from septic tanks	-	Lognormal (0.18; 0.05) [t]	Lognormal (0.18; 0.05)	Lognormal (0.18; 0.05)	Lognormal (0.18; 0.05)
P transfer coefficient in faecal sludge from biogas latrines	-	Lognormal (0.18; 0.05)	Lognormal (0.18; 0.05)	Lognormal (0.18; 0.05)	Lognormal (0.18; 0.05)
Phosphorus transfer coefficient in faecal sludge from pit latrines	-	Log. (0.3; 0.1) [t]	Log. (0.3; 0.1)	Log. (0.3; 0.1)	Log. (0.3; 0.1)
Faecal sludge emptying frequency factor *	-	Lognormal (5; 2)	Log. (5; 2)	Log. (5; 2)	Log. (5; 2)
Total food protein in food supply	g cap^{-1} day^{-1}	Normal (62.3; 6) [u]	Normal (62.3; 6)	Normal (62.3; 6)	Normal (62.3; 6)
Vegetable food protein in food supply	g cap^{-1} day^{-1}	Normal (46.2; 5) [u]	Normal (46.2; 5)	Normal (46.2; 5)	Normal (46.2; 5)
Ratio between P load in excreta and total+vegetable food protein supply	-	Normal (0.011; 0.002) [v]	Normal (0.011; 0.002)	Normal (0.011; 0.002)	Normal(0.011; 0.002)
Phosphorus load in fattener manure	kg P pig^{-1} $cycle^{-1}$	Nor. (1.5; 0.25) [i]	Nor. (1.5; 0.25)	Nor. (1.5; 0.25)	Nor. (1.5; 0.25)
Phosphorus load in sow manure (including piglets)	kg P sow^{-1} $year^{-1}$	Normal (10.5; 1) [i]	Normal (10.5; 1)	Normal (10.5; 1)	Normal (10.5; 1)
Phosphorus concentration in market wastewater	mg l^{-1}	Log. (1.7; 1) [k]	Log. (1.7; 1)	Log. (1.7; 1)	Log. (1.7; 1)
P concentration composting leachate	mg l^{-1}	Log.(75; 10) [l]	Log. (75; 10)	Log. (75; 10)	Log. (75; 10)
P content groundwater	mg l^{-1}	Log. (0.05; 0.05)	Log. (0.05; 0.05)	Log. (0.05; 0.05)	Log.(0.05; 0.05)
Atmospheric P deposition	kg P ha^{-1} $year^{-1}$	Nor. (0.9; 0.2) [w]	Nor. (0.9; 0.2)	Nor. (0.9; 0.2)	Nor. (0.9; 0.2)
Ratio between phosphorus load in urine and phosphorus load in excreta	-	Log. (0.55; 0.08) [x, y, z, aa, ab]	Log. (0.55; 0.08)	Log. (0.55; 0.08)	Log. (0.55; 0.08)
Ratio of misdiverted urine related to total urine	-	Log. (0.2; 0.05) [ac]	Log. (0.2; 0.05)	Log. (0.2; 0.05)	Log. (0.2; 0.05)
Straw ash added to pit latrines	kg cap^{-1} day^{-1}	Nor. (0.12; 0.03) [ad]	Nor. (0.12; 0.03)	Nor. (0.12; 0.03)	Nor. (0.12; 0.03)
Phosphorus content in ash	mg P $100g^{-1}$	Log. (84; 25) [ae]	Log. (84; 25)	Log. (84; 25)	Log. (84; 25)

Phosphorus load in kitchen waste	g P cap^{-1} day^{-1}	Log. (0.2; 0.1) [k, af, ag, ah, ai,]	Log. (0.2; 0.1)	Log. (0.2; 0.1)	Log. (0.2; 0.1)
Generation rate of organic market solid waste	kg m^{-2} day^{-1}	Nor. (0.7; 0.15) [k]	Nor. (0.7; 0.15)	Nor. (0.7; 0.15)	Nor. (0.7; 0.15)
Moisture content of organic solid waste	%	Log. (76; 5) [k, ah]	Log. (76; 5)	Log. (76; 5)	Log. (76; 5)
Phosphorus content in organic solid waste	%TS	Log. (0.5; 0.2) [k, ah]	Log. (0.5; 0.2)	Log. (0.5; 0.2)	Log. (0.5; 0.2)
Phosphorus load in industrial organic waste	tonnes P year^{-1}	Log. (70; 21) [b]	Log. (70; 21)	Log. (70; 21)	Log. (70; 21)
Number of piglets per sow per year	piglets sow^{-1} y^{-1}	Normal (21; 3) [i]	Normal (21; 3)	Normal (21; 3)	Normal (21; 3)
Number of breeding sows	thousand sows	Normal (37; 3)[h]	Normal (39; 3)	Normal (39; 3)	-
Phosphorus load in fattening pig manure	kg P pig^{-1} cycle^{-1}	Nor. (1.5; 0.25) [i]	Nor. (1.5; 0.25)	Nor. (1.5; 0.25)	Nor. (1.5; 0.25)
Phosphorus load in sow manure (including piglets)	kg P sow^{-1} year^{-1}	Normal (10.5; 1) [i]	Normal (10.5; 1)	Normal (10.5; 1)	Normal (10.5; 1)
Sown area of spring paddy	ha	Normal (24,260; 1000)[h]	Normal (23,000; 1000)	Normal (23,000; 1000)	Normal (23,000; 1000)
Sown area of summer paddy	ha	Normal (26,484; 1000)[h]	Normal (25,000; 1000)	Normal (25,000; 1000)	Normal (25,000; 1000)
Yield from spring paddy	tonnes ha^{-1} year^{-1}	Nor. (4.45; 0.25)[h]	Nor. (4.45; 0.25)	Nor. (4.45; 0.25)	Nor. (4.45; 0.25)
Yield from summer paddy	tonnes ha^{-1} year^{-1}	Normal (3.6; 0.25) [h]	Normal (3.6; 0.25)	Normal (3.6; 0.25)	Normal (3.6; 0.25)
Phosphorus uptake in rice straw	kg P t grain^{-1}	Normal (0.6; 0.1) [aj]	Normal (0.6; 0.1)	Normal (0.6; 0.1)	Normal (0.6; 0.1)
Irrigation water requirement (rice)	mm period^{-1}	Normal (97; 7) [r]	Normal (97; 7)	Normal (97; 7)	Normal (97; 7)
Water input in fish ponds	m^3 ha^{-1} day^{-1}	Log. (30; 15) [ak]	Log. (30; 15)	Log. (30; 15)	Log. (30; 15)
Fish pond area	ha	Nor. (3260; 50) [h]	Nor. (3400; 60)	Nor. (3400; 60)	Nor. (6800; 100)
Phosphorus concentration in urban drainage water	mg P l^{-1}	Lognormal (1.2; 1)	Log. (1.2; 1)	Log. (1.2; 1)	Log. (1.2; 1)
Organic fertiliser application rate (rice)	tonnes ha^{-1}	Normal (8; 2.5) [h and al]	Normal (8; 2.5)	Normal (24; 8)	Normal (16; 5)
Phosphorus content in organic fertiliser	kg P t^{-1}	Log. (0.4; 0.1) [al]	Log. (0.4; 0.1)	Log. (0.4; 0.1)	Log. (0.4; 0.1)
Weight gain per fattener	kg cycle^{-1}	Normal (70; 5) [i]	Normal (70; 5)	Normal (70; 5)	Normal (70; 5)
Feed conversion ratio of fattener	kg kg^{-1}	Normal (3.8; 0.5) [i]	Normal (3.8; 0.5)	Normal (3.8; 0.5)	Normal (3.8; 0.5)
Feed per sow	kg sow^{-1} year^{-1}	Normal (550; 50)[i]	Normal (550; 50)	Normal (550; 50)	Normal (550; 50)

[a] HSO (2002) and assuming 15% unregistered inhabitants; [b] Osterwalder (2006); [c] Hanoi CERWASS (2004); [d] Büsser (2006); [e] Viet Anh (2005); [f] Hanoi URENCO (2000); [g] Polprasert et al. (1981); [h] HSO (2004); [i] Ruettimann and Menzi (2001); [j] Hanoi City PC (2005); [k] Sinsupan (2004); [l] Hanoi URENCO (1998); [m] Thai (2005); [n] Ellis (2001), [o] De Bénédittis and Bertrand-Krajewski (2004), [p] Bertrand-Krajewski et al. (2005); [q] Müller (1983); [r] CROPWAT (2002); [s] Tuan (2005); [t] Montangero and Belevi (2007); [u] FAOSTAT (2004); [v] Jönsson et al. (2004); [w] FAO (2003); [x] Polprasert (1996); [y] Drangert (1998); [z] Heinss et al. (1998); [aa] Schouw et al. (2002a); [ab] GHD (2003); [ac] Jönsson and Vinneras (2003); [ad] Nghien and Calvert (2000) and assuming an ash density of 600g/l; [ae] Pasquini and Alexander (2004); [af] Diaz et al. (1996); [ag] Rytz (2001); [ah] Schouw et al. (2002b); [ai] Strauss et al. (2003); [aj] IFA (2006); [ak] FAO (1992); [al] Ha et al. (2001)

* The emptying frequency factor is a correction factor used in case the emptying frequency of on-site sanitation installations is lower or higher than average