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Towards city-wide inclusive sanitation (CWIS) modelling: modelling of faecal sludge containment/treatment processes

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OBJECTIVES

The objectives of this chapter are to:

- Promote modelling of onsite sanitation
- Familiarise readers with the basic principles of established modelling approaches applied in sewered sanitation
- Introduce ideas on how faecal sludge containment/treatment processes can be modelled using the analogy with modelling practices in sewered sanitation
- Bring sewered and onsite sanitation closer together through the integrated approach of community city-wide inclusive sanitation modelling.

6.1 BACKGROUND

The approach presented in this book is to bring urban sanitation modelling closer to city-wide inclusive sanitation (CWIS) modelling. This chapter focuses on modelling the mechanistic microbial and physico-chemical processes that take place inside a single sanitation system (to predict the faecal sludge (FS) degradation and characteristics), while an empirical approach to estimating the quantities and qualities generated in onsite sanitation systems at community or city-wide level is presented in Chapter 5.

In general there is a consensus that developments regarding urban drainage and sewerage, urban flooding, and wastewater/sewage treatment modelling have advanced to the stage that they are considered valuable and standard tools in wastewater practice. However, comparable advances in onsite (non-sewered) sanitation are lagging behind and have only made advances in the last decade. Therefore the approach in this chapter is to make as much use as possible of existing and readily accessible modelling knowledge in the wastewater and sludge treatment field (well-established modelling principles, approaches and protocols) and relate and refer, wherever meaningful, to existing modelling practices as stepping stones for the development of a roadmap for modelling onsite sanitation systems. The ultimate objective is to reach the development of a modelling tool able to predict the biological, chemical and physical characteristics of faecal sludge as a function of local and environmental factors, depending on the timescale and typical characteristics, operation and use of onsite containment/treatment technologies. For this reason, the next sections in this chapter elaborate on the basic concepts of wastewater and sludge treatment models, approaches and protocols (Henze *et al*., 2008; Rieger *et al*., 2012; Brdjanovic *et al*., 2015) to extrapolate for modelling onsite sanitation systems. Thus, three basic potential approaches are suggested to illustrate the modelling of three types of containment systems (*e.g.* a portable toilet, a pit latrine and a septic tank) either in contained or un-contained versions. This is considered the first and an essential step towards true CWIS modelling. Modelling of other CWIS components beyond these selected FSM containment/treatment technologies falls outside the scope of this chapter, as described in its concluding section.

Only recently have efforts been made to improve the understanding and description of the composition and biodegradability of faecal sludge in onsite containment and treatment systems (Elmitwalli *et al*., 2011; Lopez-Vazquez *et al*., 2013). Therefore, there is still major uncertainty about what the main biological and physico-chemical processes are that take place during the containment, emptying, transport and treatment components of the entire onsite sanitation chain, as well as what the principal underlying mechanisms are in terms of transformation processes and compounds involved, from both a spatial and a temporal perspective. In contrast, over the last three decades mathematical models have become a mature and reliable tool to support the design, optimisation, retrofit and upgrade of (activated sludge) WWTPs (Brdjanovic *et al*., 2015). However, despite numerous well-documented and published examples of successfully modelled WWTPs, examples of the application of mathematical modelling to onsite sanitation and treatment systems are rare.

Nevertheless, it should be noted that in general, human excreta (faeces and urine) are the main 'raw materials' of concern in both onsite and sewered sanitation systems. The main difference is that the fate of human excreta in sewered systems is different to that prevailing in onsite sanitation systems. As a consequence, the type of sanitation infrastructure and conditions thereof determine to a large extent the type and speed of conversion of the compounds of interest present in faeces and urine. The specific conditions characterising onsite and sewered sanitation systems have a major influence on both the composition and the quantity of sewage and faecal sludge, resulting in a different 'strength' of such streams. The characterisation and quantification of faecal sludge from septic tanks and pit latrines are elaborated in detail in chapters 2 and 5, respectively. More information on the characteristics of sewage can be found in standard sanitary engineering literature (*e.g.* Metcalf and Eddy *et al*., 2014; Henze *et al*., 2008; Chen *et al*., 2020).

It is common knowledge that the strength of sanitary flows in general is affected by the degree of dilution and the extent of the transformation process in a sanitation system as a consequence of various technical, cultural and socio-economic factors. These include: water usage and consumption, type of toilets, type of containment systems, degree of water infiltration and percolation, discharges of garbage and non-degradable materials, discharges of non-domestic waste streams, the ratio between onsite and sewered sanitation coverage, type of sewage and drainage system, management of rainwater and grey water in onsite sanitation systems, environmental conditions (*e.g.* redox, pH, temperature), hydraulics in the storage and transport components of the system, sewage and waste sludge retention time, and faecal and septic sludge retention in the containment. The dilution effect in a sewerage system is considerable, even in the case of sanitary sewers (either with or without any contribution of rainwater).

As presented in Chapter 5, the average daily production of faeces and urine of a person are 180 g and 1,500 mL, respectively, which are diluted by a large amount of relatively clean water to up to 300 or more times (Rose *et al*., 2015). Clean water is largely used for toilet flushing, evacuation of sewage from households and transport to the point of treatment and/or discharge. This decreases the concentration of the compounds of interest from the perspective of public health and environmental protection. The situation worsens in the case of combined sewers and sewers with a high infiltration rate. Other important factors are mixing and the hydraulic regimes as they play an important role in defining the environmental conditions in both sewered (*e.g.* on the flow regime in pipes and channels) and onsite sanitation settings (*e.g.* on the degree of sludge stratification in containment units and the degree of mixing and homogenisation during emptying and transport and consequent disposal), affecting the bio-chemical conversions in the system.

It is remarkable that the sanitary engineering community has been investigating activated sludge systems for more than 100 years (Jenkins *et al*., 2014) and that biological nitrification is the most studied process in wastewater treatment but that, in contrast, interest in more fundamental research on onsite sanitation systems has only gained momentum in recent years. This is even more surprising given that the initial 'raw material' of concern in both onsite and sewered sanitation systems is essentially the same. The failure to distinguish the principal differences between raw sewage, faecal sludge and septic sludge has, on occasion, led to the collapse of existing wastewater treatment plants because they have not been designed to receive high(er) faecal sludge loads (Lopez-Vazquez *et al*., 2013). Without doubt, there is still a lack of fundamental understanding and scientific evidence of the complex processes taking place in onsite sanitation systems across the world, including latrines as the most common onsite containment unit. However, thanks to the fact that sanitation has had a prominent focus in both UN Millennium Development Goals (MDGs) and is increasingly prominent in the current UN Sustainable Development Goals (SDGs, United Nations, 2015), the interest of both the academic and professional communities in sanitation has increased tremendously over the last decade, in recognition of the fact that onsite sanitation has to be approached with deeper insight, advanced knowledge and greater confidence. Therefore, the authors believe that this chapter will improve the understanding of the dominant microbial and physico-chemical processes that take place in onsite sanitation systems. This understanding should be based on the principles, fundamentals and proven practice documented by researchers, modellers and practitioners dealing with sewers, activated sludge systems and sewage sludge, that can be used as a basis to define the approaches and steps required for modelling onsite containment systems in order to estimate the volumes and characteristics of the faecal sludge generated in different sanitation systems. It will ultimately contribute to the development of a modelling framework that could potentially be used to improve the design and exploitation of onsite and also sewered sanitation systems in the future.

The expected benefits of setting out the basis for modelling onsite sanitation systems are: (*i*) to improve practical understanding of onsite sanitation systems, (*ii*) to increase confidence in the

determination of the main faecal sludge characteristics and fractions, (*iii*) to deepen fundamental understanding of the dominant/prevailing biological and physico-chemical processes that take place in onsite sanitation systems, and (*iv*) to help to initiate a community of practice on onsite sanitation modelling.

It is important to highlight that although there are several mathematical models capable of satisfactorily describing carbon, nitrogen and phosphorus removal processes from sewage, pathogen removal has been overlooked by the mainstream modelling community. However, since most onsite sanitation systems are located in low and middle income countries where billions of people have no basic sanitation provision, it is essential to contribute to the prevention of the spread of waterborne diseases and therefore to prevent contact of people with pathogens through the control of contamination pathways and pathogen-removal mechanisms. This is an obvious reason and an important challenge to develop and promote enhanced pathogen removal (or inactivation) practices and approaches supported by mathematical modelling and linked to the transformations of other compounds (*e.g.* organics, nitrogen and phosphorus). Two-directional synergy between the two sanitation fields, in a spirit of CWIS, is useful and recommended given the fact that, for instance, by promoting the generation of inactivation agents during faecal sludge treatment, pathogen reduction and inactivation can be achieved (Nordin *et al*., 2009; Fidjeland *et al.*, 2013; Anderson *et al.,* 2015).

Similarly to sewage-based modelling, onsite sanitation modelling can have the potential to become a basis or a tool to improve the management and operation of sanitation facilities in onsite settings because, for example, the actual removal capacity, volume and solids accumulation in onsite systems could be better predicted and improved, also enabling better emptying practices (Bhagwan *et al*., 2008). Recent large faecal sludge characterisation efforts in Sub-Saharan Africa and South Asia and approaches to track material flows (well-established in the wastewater field in the form of mass balances) and represented by Shit Flow Diagrams (SFDs, Peal *et al*., 2020) are clearly important building blocks of the foundation needed for onsite sanitation modelling. Similarly to the latest trends in wastewater treatment, the quantification and prediction of the transformation processes of faecal sludge may make it possible to replicate developments such as 'WWTP - an energy factory' and 'energy-neutral WWTP' in some way within the onsite sanitation field. As such, despite the intrinsic complexities and drawbacks, it becomes very important to promote modelling of sanitation systems within the framework of a CWIS approach (World Bank, 2019; Löthi and Narayan, 2019), to contribute to the development of the sanitation value and service chain management in an integrated and holistic way.

6.2 INTRODUCTION TO MODELLING – LEARNING FROM ACTIVATED SLUDGE MODELS

6.2.1 What is a model?

A model can be defined as a purposeful representation or description (often simplified) of a system of interest (Ubisi *et al.*, 1997). This consequently means that a model never exactly reflects the reality. So, the question 'Can (does) this model describe a process occurring in an onsite containment system?' is pointless without a definition of what (which) part(s) of an onsite containment system the model should describe. One never develops a model that describes every detail of the process. Models are a simplification of reality that describe that part of reality that is relevant to understand and to deal with (Van Loosdrecht *et al.*, 2015). It is also important to note that a mathematical model can only be successful if it fulfils the expectations that people have of it. From the perspective of time, a model can be developed to describe frozen-state, dynamic-state or steady-state conditions. Frozen-state conditions are those that do change over time, but not in the time interval that one is interested in or dealing with. Often, dynamic-state conditions are the ones that deserve special attention to describe the variations that occur as a function of time. For instance, the concentrations of organic matter, nitrogenous and phosphorus compounds in the influent will vary during the day, the

concentration of ammonia in the effluent will vary over time, concentration of nitrate will vary in the activated sludge recators etc. Nevertheless, the concentrations of these compounds in anaerobic sludge digesters (which nowadays with an increased interest in energy and resources recovery are often found as intrinsic components in sewage treatment plants) (Batstone *et al*., 2014) scarcely vary within a day. One of the reasons is that the hydraulic retention time (HRT) of sludge digesters is usually around 20-30 days and, thus, the characteristics change in time intervals of two to three weeks. As a consequence, the variations or fluctuations in sludge digesters, with regard to the daily dynamics of interest, are therefore assumed to be in a kind of frozen state. The analogy can be drawn with some onsite containment systems that are also less sensitive to daily variations in the load and are based on anaerobic digestion (*e.g.* septic tanks or pit latrines that are not often emptied). Moreover, some other processes occur so fast that they are assumed to be, under the usually applied timescales of a study, under steady-state or equilibrium conditions. An example of such processes are the precipitation processes that occur almost instantaneously (in a few seconds). The speed at which these processes occur is so fast that they do not have to be described in a dynamic way, so they are assumed to be in equilibrium or completed. As such, one of the first considerations is to define what the processes of interest are, the relevant timeframe for their description, an assessment of their dynamics, and an accurate description of those processes that are time-variable within the timeframe of concern. Therefore, the aspect of time is the first major issue in trying to simplify the reality. The recommended approach is to consider the time constants and select those processes that have the dynamics in the order of time constants that one is interested in.

The second relevant issue for modelling is space resolution. One can theoretically make a model that describes every square inch of the process tanks, reactors or section of a sanitation system. However, one needs to realise and define whether such a detailed description is strictly necessary. The answer can be found, once again, in the purpose of the model. In order to describe the concentration gradients of the relevant components in the process tanks, units or reactors, one should determine the scale size that is most appropriate. On a different scale, there is a gradient of concentrations inside the bacterial agglomerations, biofilms, and accumulation of solids that theoretically can also be described by a model. Again, the situation may be different in onsite containment systems (such as pit latrines or septic tanks) where stratification, water content of the sludge, and limited or no mixing may all have a major influence on the choices made. Therefore, one needs to assess whether they are sufficiently relevant to be taken into account.

The next step in modelling is the relevant level of detail in a microbial model. In activated sludge modelling, the closest modelling parallel, there are basically three approaches (Van Loosdrecht *et al*., 2007): *(i*) the traditional 'black-box' approach, (*ii*) the 'grey-box' approach, and (*iii*) the 'glass-box' approach. Over the years, the black-box approach has been shown to be reliable enough for design purposes, even though it does not provide information about the sludge composition. If one is interested in refining the design and operation of the plant, grey-box models (such as Activated Sludge Model No. 1 - ASM1) split the sludge into relevant fractions composed of the compounds of interest (such as biodegradable and unbiodegradable, soluble and particulate fractions) and microbial biomass (such as ordinary heterotrophic organisms, nitrifying organisms, phosphate-removing organisms, among others). This approach allows modeller to take into consideration different functional aspects of the microbial communities present in the sludge and incorporate them in the model. 'Glass-box' models, such as the metabolic models initially developed for enhanced biological phosphorus removal (EBPR) by Smolders *et al.* (1995), Kuba *et al.* (1996), Murnleitner *et al.* (1997) and for the first time applied at a full-scale WWTP by Van Veldhuizen *et al.* (1999) and Brdjanovic *et al*. (2000), provide a good description of the metabolic routes that take place inside the organisms, almost reaching a 'glass-box' modelling approach. This more complex and detailed level has been shown to be necessary to secure a satisfactory description of phosphorus-removing systems, but it is by no means essential to

describe all the biological processes. Therefore, the preference for a black-, grey- or glass-box modelling approach depends on the purpose and application of the model, also in the context of onsite sanitation systems.

Furthermore, two types of mathematical models exist: empirical and mechanistic models. An empirical model is based on the recognition of the parameters that seem to be essential to describe the behavioural patterns of interest, and linking these through empirical relationships established by observation (*e.g.* mathematical regressions to find any dependence between the effluent characteristics and the influent concentrations or environmental conditions such as temperature). As such, in empirical models, the mechanisms and/or processes operating and governing the conversions that occur in the system are not known and are often ignored. Empirical models can be considered to be an example of a classical black-box modelling approach. In contrast, a mechanistic model is based on a particular conceptualisation of the biological/physical mechanisms governing the system. The degree and level of understanding of the biological and chemical processes occurring in the system will determine the complexity of a mechanistic model. As such, since mechanistic models have a conceptual basis, they tend to be more reliable than empirical models. Moreover, empirical models are naturally restricted by the boundaries used to develop the model itself (such as the wastewater or faecal sludge characteristics and system parameters), allowing only certain interpolation. On the other hand, because mechanistic models are conceptually-based, they can be not only interpolated but also extrapolated. Nevertheless, one should not forget that all models need to be rigorously and properly calibrated and verified. In addition, the boundary conditions of application of every model should also be firmly delineated. Historically, and based on how they have been developed and evolved, mechanistic models have been shown to have a greater potential for application in the sanitary engineering field, deserving special attention and interest compared to empirical models.

To set up a mechanistic model, a conceptual model needs to be defined describing the processes of interest occurring within a system and the compounds subject to the transformations and conversions to be described by the processes. Furthermore, the interactions and interlinks between the processes and compounds should also be delineated. Thereafter, a mechanistic model can be developed by formulating the mathematical expressions that describe the stoichiometric relationships and kinetic rates of the processes and their compounds. Strictly speaking, the model should not include all the processes that take place within a system but only those that are significant to meet the expectations raised by the model. To develop a model that includes all the possible processes and their interactions is not feasible, since it would lead to a very complex model that would not completely describe the phenomenon. An example of such a practice is the level of organisation: rather than model every microbial population (for which microbial identification and enumeration techniques may not even be fully and reliably developed) microorganisms are grouped as single entities or groups of 'surrogate' organisms that fulfil or perform a defined function, namely: ordinary heterotrophic organisms (OHO) that carry out the aerobic removal of organics on the upper layers of an onsite containment system that are exposed to air, or anaerobic organisms (ANO) that perform the removal of organics in the deeper layers of the same onsite sanitation system where oxygen is absent. The single entities or surrogate groups of organisms are modelled with a defined set of characteristics and behaviour to describe their prime function within the system. These characteristics will not reflect the particular or specific characteristics of each individual microorganism, but their main function or process of interest that, as a whole, will provide a satisfactory description of the main role of the group in the system. Consequently, the actual overall effect of modelling the group reflects the cumulative net effect of the individual contribution of each microorganism. The advantage of this approach is that it decreases the level of complexity since less information is required for the development, calibration and validation of the model. Usually, most of the information and parameters that are

incorporated are of a biochemical or microbiological nature. Also, the more complete, the better the description. Nevertheless, this additional information should be incorporated to the minimum required level where the key processes that govern or describe the response of a system are identified. This is also because detailed microbiological and biochemical information is usually needed (Ubisi *et al.*, 1997) and, even more importantly, data from onsite sanitation systems is often subject to considerable fluctuations and levels of uncertainty (Brouckaert *et al.*, 2013). In this regard, more methods are needed to quantify uncertainty and its sources (Sin *et al*., 2005; Belia *et al.*, 2009; Benedetti *et al*., 2010; Flores-Alsina *et al*., 2012) in onsite sanitation systems. The adaptation to onsite sanitation systems of the findings and developments of the IWA Task Group on Design and Operations Uncertainty (DOUT) (Sin *et al*., 2005; Belia *et al.*, 2009; Flores-Alsina *et al*., 2012) can be used to carry out uncertainty evaluations and contribute to defining the minimum levels of complexity and data required to describe the operation and performance of faecal sludge technologies.

The objectives that the model needs to fulfil will determine the parameters that need to be considered based on the defined level of organisation. Generally, two different types of models are developed: steady-state and dynamic models. Steady-state models are simpler since they usually have constant or steady constant flows and loads. Dynamic models are more complex because they tend to have variable or varying flows and loads. Steady-state models are oriented to determining the most important design parameters and therefore are good for design. Dynamic models are useful to predict the time-dependent response of a treatment system.

6.2.2 Modelling basics

6.2.2.1 Model building

Mathematical models can provide a quantitative description of the systems of study and, therefore, are widely applied. Mathematical expressions are used to describe the stoichiometric reactions and the kinetic rates at which the conversions of the parameters occur (usually as a function of time). To provide the required predictions, the mathematical formulations are included with the procedures needed to find their solutions within the boundaries defined by the structure of the model and that of the system (such as temperature and mixing conditions). Mathematical models are not developed in isolation but evolve in close interaction with conceptual and physical models (*e.g.* laboratory-scale or pilot-scale reactors) (Ubisi *et al.*, 1997).

For example, to develop a mathematical model that describes the wastewater (or faecal sludge) conversion processes that take place in sewered or onsite systems, at least four components are needed: (*i*) influent or input characteristics, (*ii*) balance equations, (*iii*) kinetic process rates, and (*iv*) transport processes, as described below.

Influent or input characteristics

An adequate and reliable determination of the influent or input characteristics is vital in order to obtain a satisfactory description of the process conversions and of the actual impact and response of the system. Bearing in mind the objectives to be met, the level that existing models have reached implies carrying out not only a thorough characterisation during a representative period of time, but also a fractionation of the compounds of interest. The characterisation should look into those parameters that better illustrate the strength of the medium (*e.g.* BOD, total COD, soluble COD, total nitrogen, ammonia, among others). Also, the characterisation must include the determination of limiting compounds (whose absence can limit the conversion processes) and inhibiting or toxic compounds (whose presence can slow down or even prevent the (bio-) degradation or conversion processes). An example of limiting compounds can be oxygen for the aerobic removal of organic matter, and an example of inhibiting or toxic compounds can be ammonia or hydrogen sulphide for the anaerobic removal of organic matter. Regarding the fractionation of the main compounds of interest (at least of organic matter in terms of COD), this should be carried out in relation to the potential conversions that are closely related to their physico-chemical and (bio-) degradation properties, under the prevailing redox conditions (generally, aerobic or anaerobic). In this regard, most available protocols focus on the

determination of the soluble and particulate fractions, and to what extent these soluble and particulate fractions are (bio-)degradable or not, within the boundaries of the conversion process in question (Hulsbeek *et al.,* 2002*;* Van Loosdrecht *et al*., 2015). The determination of the characteristics and fractionation(s) is in general an essential modelling step (also in the cases of faecal and septic sludge), since it contains the main input data of the model and, as expected, will define the success of the description of the conversion processes. Moreover, its correct determination will ease the calibration and validation process (Brdjanovic *et al*., 2015).

Balance equations

Balance equations are necessary to describe the biological and chemical conversion processes of interest. These processes lead to the consumption of reactants and the generation of products. Often, a product generated by one reaction or conversion process can be the reactant of one or more subsequent processes. Consequently, the concentrations of certain compounds, or parameters in a reactor or system, will change over time. However, when a system or model reaches steady-state conditions, the concentrations are stable and therefore no longer change.

Kinetic process rates

Each reaction has its own rate equation. The rate equations specify the rate at which certain reactants are converted into their products. The kinetic rate expressions can be either substrate-based or growth-based. They range from zero order to second order equations (*e.g.* $r = k$, $r = kC$ to $r = kC^2$, where r is the reaction, k is the kinetic rate and C is the concentration of the component converted during the reaction) (Metcalf and Eddy *et al.,* 2014). In waste(water) conversion models, the most common process rate equations used are the saturation equations defined by, for example, the widely used empirical Monod-type expressions (*e.g.* $r = kC/(K+C)$ where K is the half-saturation concentration of the component converted during the reaction. Such an expression allows us to describe the process rates as a function of the availability of substrate in the systems and reactors (in this case, C). It is also common to use Monod-type equations as switching functions (in the form of $r = K/K + C_I$) to describe the inhibitory effects caused by a toxic or inhibiting compound (C_I) that slows down a reaction process and at high concentrations can even stop it.

Transport processes

Together with the stoichiometric equations and kinetic process rates, transport processes also affect the changes in concentrations in a reactor over time, because the local concentrations observed in a process unit or reactor (besides being affected by the conversion processes whose rates are usually dependent on the local concentrations themselves) are also subject to the transport of reactive compounds between process units. The transport processes can be convective or diffusive. Convective transport processes are commonly used to describe the transport of liquids (directly linked to the hydraulic behaviour of a plant, such as the conduction of a wastewater stream from one tank to another), whereas diffusive transport processes are used to describe the transfer of gases between phases (for instance, to describe the diffusion of oxygen from the atmosphere into a liquid contained in an open reservoir). Thus, transport processes are another key component of a model of a physical nature and must also be carefully determined.

6.2.2.2 General activated sludge model set-up

The different influent or characteristics of the inputs, balance equations, kinetic process rates, and transport processes are the main components of a model. They need to be grouped following a defined framework to provide an adequate representation of the onsite containment system dependent on the objectives pursued by the modelling study. First, the stoichiometric equations that define the main conversion processes of interest need to be incorporated. From a conservation perspective, they need to be mass-balanced to comply with the conservation principles (all inputs should equal all outputs) in terms of loads (*e.g.* carbon, phosphorus) and charges (*e.g.* for nitrogen compounds, alkalinity). Together with their correspondent kinetic process rates, these balanced equations are the main core of the conversion models. Over the years, different research groups and groups of practice have developed extensive aerobic and anaerobic models

that present, in a structured manner, the main stoichiometric and balanced conversion processes, as well as their corresponding stoichiometric and kinetic rates and parameters. Among others, with regard to the conversions of organic matter, examples of such developments are the activated sludge and anaerobic digestion models (ASM1 and ADM1, respectively) to describe aerobic- and anaerobic-driven organic matter conversion processes, respectively. The use and application of certain models (for instance, either an ASM-type or ADM1) depends on the objective of the modelling study. Consequently, the most suitable model(s) need(s) to be selected to model either the system or certain process units with one model type (*e.g.* aerobic phases with an ASM-type model) or with another (*e.g.* anaerobic phases with ADM1).

Once the model has been selected, measurable input parameters and fractionations need to be determined as a function of the selected model. In this regard, ASM1 requires very basic characterisations and fractionations (composed of only four COD parameters as a function of their complexity and biodegradability to describe the COD loads) (Henze *et al*., 1987). On the other hand, ADM1 demands a very thorough and extensive characterisation and fractionation that requires the determination of carbohydrate and lipid concentrations (among other compounds) in the influent (Batstone *et al.,* 2002). After the determination of the corresponding wastewater characteristics and fractionations, they are transformed into an influent vector, becoming the main input of the model.

The transport processes in the FSM unit need to be defined based on the transport (flow or flux) of the main streams or discharges through the treatment system. Initially, the system can be modelled hydraulically, describing the main zones/reactor compartments of the system. An approach is recommended in which each process unit is modelled individually considering its hydraulic behaviour (whether it is a completely-stirred tank or a plug-flow reactor) and redox conditions (aerobic or anaerobic). The process units may be further split or divided into compartments to mimic the dominant or prevailing conditions. For instance, a process unit with plug-flow hydraulic behaviour can be represented by a defined number of completely-stirred tank reactors (CSTR) in series. This practice is common to ease the modelling process (Volcke *et al*., 2006). Also, one process unit can be split into different compartments to represent the existence or generation of different redox conditions (such as anaerobic or anoxic dead zones in aerobic units due to uneven mixing or aeration conditions). In all the aforementioned conditions, the transport of flows and the concentrations of the compounds of interest between process units and their compartments can be described with convective transport expressions based on the actual hydraulic configuration of the treatment system. With regard to the transport of gases, diffusive transport expressions are commonly applied. This enables the diffusion of oxygen into the process units to be assessed as well as the gas emissions from the conversion processes.

It should be noted that neither the ASM nor the ADM families of models include pathogen removal. Therefore, pathogen removal/inactivation modelling and its integration with other models is addressed in section 6.3.7.3.

The overall model of a system can be generated by compiling the influent characterisation model (or influent vector), the process conversion model (containing the stoichiometric and kinetic components) and the process flow model. The process flow model can be composed of individual units and their phases or sub-units connected by a state vector that includes the corresponding convective and diffusive transport expressions, as required. The overall model is usually solved numerically to compute the concentration of each compound included in the model as a function of time. Every compound entering into the treatment system and consequently into each process unit, reactor or compartment should be converted, exchanged with the gas phase, or leave with the effluent. For example, a schematic representation of the model of a sewer-based system, an activated sludge wastewater treatment plant, is presented in Figure 6.1. It is composed of four units or phases modelled as a continuous stirred-tank reactor

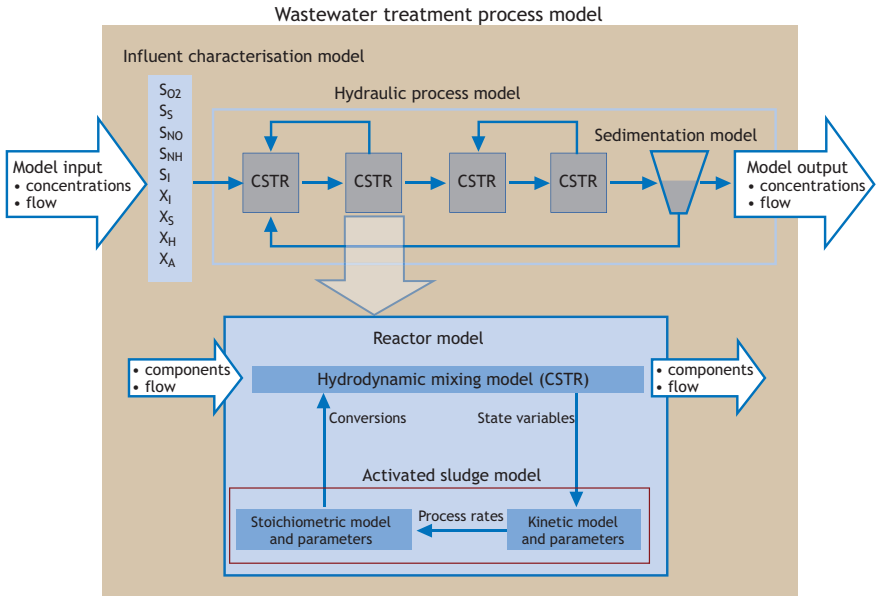


Figure 6.1 Schematic representation of an activated sludge wastewater treatment plant (modified from Meijer, 2004).

(CSTR), interconnected to simulate the potential recycle and return of flows between the tanks. The feed or influent is received or discharged into the first unit before continuing to flow to the next units. The retention and/or accumulation of solids (the solid-liquid separation process) is modelled as a sedimentation model or settling unit before discharging the effluent. In particular, every single unit has both a hydrodynamic model and a process conversion model to describe the transport and conversion processes of the compounds of interest.

6.2.2.3 The matrix notation

Balance equations are used to describe the conversions of the individual compounds of interest depending on the objectives and purpose of the modelling study. Due to the number of relevant compounds, their associated conversions and the dependencies between balance equations (in which the product of a balance equation can become the reactant of other equations whose products can be the reactants of other processes, and even of the previous processes, and so forth), in 1987 the IAWQ Task Group on 'Mathematical modelling of wastewater treatment' (Henze *et al.*, 1987) recommended and adopted the Peterson matrix notation (Peterson *et al.*, 1965), afterwards renamed the 'Gujer matrix', for model presentation. This format facilitates a clear and unambiguous presentation of the compounds and processes and their interactions in a simple and compact manner. Moreover, this format allows a direct comparison between different models, and facilitates the transfer of the expressions into a computer program or modelling simulator. The matrix is presented by a number of columns and rows, in which the columns are used to display the compounds of interest and the rows the processes to which the compounds are subject to, either as reactants or products. Table 6.1 presents an example of a simplified stoichiometric matrix that describes the aerobic removal of readily biodegradable organics (S_S with the stoichiometric coefficient $-1/Y_H$) by the aerobic growth of ordinary heterotrophic organisms (X_H with the coefficient +1) linked to oxygen consumption (S_O with the stoichiometric coefficient $-1/Y_H+1$). A negative coefficient indicates that a component is consumed whereas a positive coefficient indicates that the component is produced or generated. The process rate of the aerobic growth reaction is $\mu_H^{MAX} \cdot (S_S/(K_S+S_S)) \cdot X_H$). The example also includes the lysis or decay process of the ordinary

Table 6.1 Example of a simplified stoichiometric matrix for activated sludge modelling (Henze *et al.*, 1987).

Components i	1: S_O	2: S_S	3: X_H	Process rate equation ρ_j
List of processes j				
Aerobic growth	$-\frac{1}{Y_H}+1$	$-\frac{1}{Y_H}$	+1	$\mu_H^{max} \cdot \frac{S_S}{K_S+S_S} \cdot X_H$
Lysis		+1	–1	$b_H \cdot X_H$
Observed transformation rate r_i	$r_i=\sum_j v_{j,i} \cdot \rho_j$ $[M_i L^{-3} T^{-1}]$			
Definition of stoichiometric parameters: Y_H Heterotrophic yield coefficient $[M_H M_S^{-1}]$	Dissolved oxygen (O_2)	Dissolved organic substrate (COD)	Heterotrophic biomass (COD)	Definition of kinetic parameters: μ_H^{max} Maximum specific growth rate $[T^{-1}]$ K_S Saturation coefficient for substrate $[M_{COD} L^{-3}]$ b_H Rate constant for decay $[T^{-1}]$

heterotrophic organism (OHO) biomass (negative coefficient -1) that results in the generation of readily biodegradable organics (S_S with the positive coefficient +1) with a process kinetic rate $b_H \cdot X_H$ (Henze *et al.*, 1987). This example also illustrates the potential interconnections between components in which the product of the first reaction (the heterotrophic biomass, X_H, generated during the aerobic growth process) becomes the reactant of the second reaction (in the lysis process) and, consequently, the product of the second reaction (S_S) is the reactant of the first reaction.

In the previous example, all the units are expressed in terms of COD equivalents and the continuity and, therefore, conservation principles need to be met. These can be assessed by moving across any row in the matrix, summing up all the coefficients whose net sum should be zero. The previous example illustrates how a matrix can be used to summarise and represent complex interactions between compounds and processes in a relatively simplified manner, justifying why the matrix notation is commonly used in mathematical modelling of wastewater treatment systems. It is strongly recommended that matrix notation is used in modelling of onsite sanitation systems, following any necessary adaptation.

6.2.2.4 Wastewater treatment models

As previously described, different extensive aerobic and anaerobic models have been developed over the years to model sewered sanitation systems, and in particular activated sludge systems. The family of mathematical models developed under the leadership of the International Water Association (IWA) includes the most applied models in the field of wastewater treatment. These include the ASM models nos. 1, 2, 2d and 3 (Henze *et al.*, 2000) and ADM1 (Batstone *et al.*, 2002. Also, previous versions that have contributed to the development of the IWA models can be found, such as the UCTOLD or the UCTPHO models (Dold *et al.*, 1981,; Wentzel *et al.*, 1988, 1989a, 1989b), models with a similar basis developed in parallel (Barker and Dold, 1997) or modified or expanded versions of the IWA models (such as the TUDelft model, or the ASM3-Bio-P model) (Meijer, 2004; Rieger *et al.*, 2001). However, in spite of the development of different anaerobic models since the late 1970s (Donoso-Bravo *et al.*, 2011), IWA ADM1 (Batstone *et al.*, 2002) is still the most commonly applied anaerobic treatment model. One important reason is that its core model structure with different adaptations, modifications and extensions (Donoso-Bravo *et al.*, 2011) has proven capable of describing several wastewater and solid waste conversion processes (Kythreotou *et al.*, 2014; Batstone *et al.*, 2015). Furthermore, with the use of

suitable interfaces, coupling ASM-types with the ADM1 model has become possible for plant-wide modelling purposes (Mithaiwala *et al*., 2005; Rosen *et al*., 2006; Volcke *et al*., 2006; Alex *et al*., 2008; Nopens *et al*., 2009) with the aim of optimising the operation of wastewater treatment plants and for resource recovery purposes. For the implementation of the models, different general purpose simulators are available ranging from open-access simulators such as Aquasim, ASIM[1] or STOAT[2] to proprietary software simulators such as MatLab™/Simulink™[3]. In parallel, different initiatives have led to the development of more comprehensive models that couple aerobic and anaerobic processes. They often belong to more advanced commercial software packages and include BioWin[4], GPS-X[5], SIMBA[6], SUMO[7], and WEST[8]. Some of these comprehensive models have been incorporated in simulators that bring additional advantages. For example, they offer user-friendly interfaces to build process-flow diagrams of sewered sanitation systems, to describe more easily the key chemical and precipitation processes, or to estimate specific operating conditions that can lead to process inhibition due to the presence or accumulation of certain compounds (*e.g.* sulphide, excessive ammonia or nitrite accumulation). All the aforementioned models have defined model structures to describe certain conversion processes and therefore meet specific modelling objectives. Thus, a key decision in the modelling process is to select the model that is most suitable for the required modelling needs. This selection is usually carried out by considering the main conversion processes that take place in the system to be modelled and those that each model can describe. Consequently, the model whose conversion processes are identical or the closest to those governing the system under study can be selected. Excluding models that belong to or are part of proprietary simulators or software packages, Table 6.2 presents an overview of some selected (open-access) models developed for wastewater treatment with specific emphasis on the main conversion processes that they can describe. For modelling onsite sanitation systems, certain processes can probably be excluded (such as nitrification, denitrification and enhanced biological phosphorus removal (EBPR), which require the presence of oxygen prior to, during, or after each of these processes) bearing in mind that most of the conditions prevailing in onsite containment units tend to be anaerobic (due to the absence of aeration systems) or that they are micro-aerophilic (in the upper layers of the systems) (Bakare *et al*., 2012). As such, to describe the conversion processes occurring in onsite sanitation systems, ADM1 appears to be an essential model coupled with ASM1 or ASM3 to describe the marginal aerobic processes.

6.2.2.5 Modelling protocols

As described previously, different mathematical models have been developed and extensively applied to model several types of aerobic and anaerobic wastewater treatment systems. For this purpose, each model requires to be calibrated for each case study. As such and since different research groups, groups of practice and experts, companies and institutions have been involved in the implementation of modelling studies in different regions, several calibration models have been developed involving different methodologies and approaches (Hulsbeek *et al.,* 2002*;* Vanrolleghem *et al.*, 2003; Sin *et al*., 2005). Among them, four calibration protocols have become most popular (Sin *et al*., 2005): (*i*) the BIOMATH calibration protocol (Vanrolleghem *et al.*, 2003), (*ii*) the HSG guidelines (Langergraber *et al.*, 2004), (*iii*) the WERF protocol for modelling calibration (Melcer *et al.*, 2003) and, (*iv*) the STOWA calibration protocol (Hulsbeek *et al.*, 2002; Roeleveld *et al.*, 2002). Despite the advantages and disadvantages of each protocol, all of them have a similar structure.

[1] www.eawag.ch
[2] www.wrcplc.co.uk
[3] www.mathworks.com
[4] www.envirosim.com
[5] www.hydromantis.com
[6] www.ifak.eu/content/simba-sharp-water
[7] www.dynamita.com
[8] www.mikepoweredbydhi.com/products/west

Table 6.2 Overview of selected mathematical models commonly applied to model sewered treatment systems (modified from Gernaey *et al.*, 2004).

Model	Aerobic organic matter removal	Nitrification	Denitrification	EBPR	Chemical P removal	Hydrolysis	Fermentation	Acetogenesis	Methanogenesis	Reference
ASM1	●	●	●			●				Henze *et al*. (1987)
UCTOLD	●	●	●			●				Dold *et al*. (1981,)
ASM3	●	●	●			●				Gujer *et al*. (1999)
UCTPHO	●	●	●	●		●	●			Wentzel *et al*. (1988, 1989a, 1989b)
ASM2	●	●	●	●	●	●	●			Gujer *et al*. (1995)
ASM2d	●	●	●	●	●	●	●			Henze *et al.* (1999)
B&D	●	●	●	●		●	●			Barker and Dold (1997)
TUDP	●	●	●	●		●	●			Meijer (2004)
ASM3-BioP	●	●	●	●		●				Rieger *et al*. (2001)
ADM1						●	●	●	●	Batstone *et al*. (2002)

Sin *et al*. (2005) carried out a thorough SWOT (Strengths, Weaknesses, Opportunities and Threats) analysis of the calibration protocols previously listed (BIOMATH, WERF, HSG and STOWA). Overall, they concluded that all of them are suitable and reliable; the BIOMATH calibration protocol is the most sophisticated (with regard to its level of detail and thorough characterisation and calibration procedures), the HSG is the most systematic (concerning the calibration steps), the WERF is the most detailed with regard to the experimental methods needed for influent characterisation and fractionation (including a summarised number of calibration studies, which is attractive for inexperienced modellers and consultants), and the STOWA calibration protocol, which is the most straightforward, practical and easy to implement. In particular, the STOWA protocol can be useful for inexperienced modellers and practitioners, since it also gathers and summarises the experience earned through several modelling studies (Roeleveld *et al*., 2002). Therefore, since the most commonly applied modelling protocols share and follow, to some extent, similar concepts and principles. The STOWA calibration protocol will be briefly presented in this section and the main steps discussed from an faecal sludge modelling perspective. Figure 6.2 presents a flow diagram illustrating the main steps of the STOWA calibration protocol and their inter-relations (Hulsbeek *et al*., 2002). These are discussed in more detail below.

Formulation of objectives

The definition of the main purpose and objectives is essential to define the scope of the study, its relevance, and also its boundaries. The objectives define whether the modelling study will be carried out to select a (future) design, to optimise an existing design or to develop (improved) strategies to operate existing or future sanitation systems. This will influence the model extension and complexity, and also the required modelling activities, such as the length and frequency of the sampling campaigns and the type and number of operating and analytical parameters to be determined and analysed.

Process description

Depending on the objectives of the study, the process can be described by defining the process components of relevance and identifying the general plant layout and configuration. It is essential to include and define all the inflows, internal flows and outflows from the system (*e.g.* influent, feeds, internal recirculations, effluent, infiltration and percolation flows, whenever applicable).

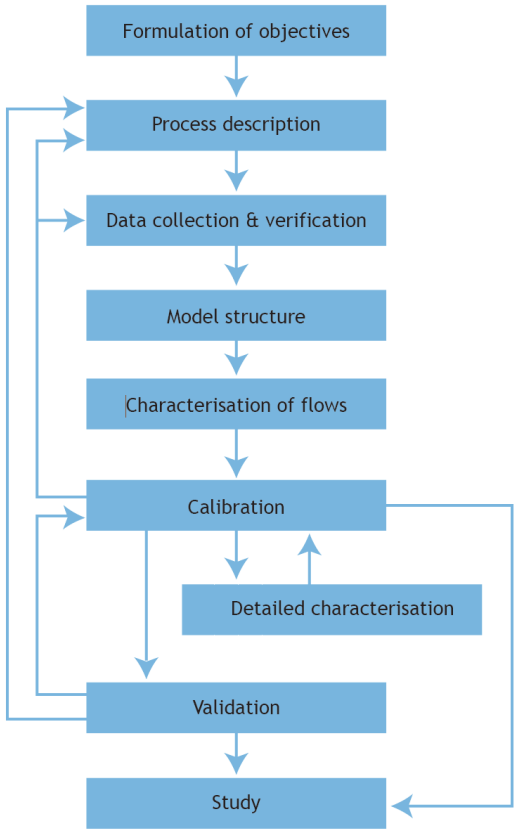


Figure 6.2 Main structure of the STOWA calibration protocol (Hulsbeek *et al.*, 2002).

Data collection and verification

The collection of data is essential in conducting a comprehensive survey of the system under study. In this step, the composition and volume of all the flows going through all the process components need to be defined. If available, data can be collected from (previous) periodic sampling and monitoring programs. This data can be useful to start to define the characteristics and composition of the flows. Furthermore, this preliminary information can be used to run preliminary simulations (after selecting a model) and use them to design an appropriate and more detailed sampling and monitoring campaign to complete the data required for modelling. The concentrations that show the highest variations at certain points may need to be evaluated in detail. It is highly recommended to evaluate the quality of the data collected to find potential gaps and to correct any potential inconsistency. For this purpose, it is strongly advised to conduct water and mass balances on the suspended solids, COD, nitrogen and phosphorus (Meijer, 2004). Depending on the outcomes of the data quality assurance (*e.g.* if the mass balances do not close), additional sampling and monitoring campaigns will be needed to take this into account to complete and/or correct the required data.

Model structure

The structure of the model will be initially defined based on the process description. First, the model will need to be set up based on the hydraulics or transportation processes of the FSM unit, defining each process component. This means that the number of tanks, the compartments of the tanks, redox conditions, and solid-liquid separation compartments will need to be defined. The redox conditions will not only indicate whether a tank or stage is anaerobic or aerobic but also if the redox gradients prevailing in the system may indicate that one single tank should be modelled as a series of aerobic or anaerobic compartments. To set up a proper compartmentation, it is recommended to measure the dissolved oxygen concentration and redox conditions in a vertical and horizontal direction in all the tanks and their compartments. Based on the prevailing or dominant processes conversions, a process model needs to be selected among those available in literature (*e.g.* ASM-type, ADM1).

Characterisation of flows

First (if available), using historical data or specific measurements, the main inputs and flows can be characterised. Depending on the configuration of the system, these need to include the influent, effluent, and the internal and recirculation flows. If there is no data available or certain data points are missing, a sampling campaign needs to be conducted. If the model will be used to select a design, daily average concentrations for three days and the variations in the flow patterns may be enough. However, for process optimisation and control strategies, samples may need to be collected periodically every 2-4 hours over a period of three to seven days at several critical points along the system (*e.g.* not only at the feed or influent and effluent but also at the interfaces

between the tanks and compartments). All the data gathered and collected needs to be checked for consistency (*e.g.* performing water and mass balances).

Calibration

Once the data have been checked for consistency and quality assurance, the first simulations can be executed and the model calibrated using the available data. If the description of the performance of the plant shows that a major adjustment is needed (*e.g.* if in order to describe the data or measurements a large adjustment of the kinetic parameters is required), the model structure will probably need to be revised as well as the mass balances and data collection. Based on the experience drawn from modelling activated sludge systems, it is recommended to first model and calibrate the sludge production, followed by the process which is kinetically most sensitive, and afterwards the rest of the kinetic processes. If the process performance and effluent quality are not well predicted, a sensitivity analysis can be conducted to assess which parameters have the strongest impact. In this regard and at this stage, different approaches can be applied to quantify the level of uncertainty and its sources and to assess in more detail their impact to define additional sampling and monitoring criteria (Belia *et al*., 2009; Flores-Alsina *et al*., 2012). Following an iterative step-wise process, the model could be calibrated by adjusting the least possible number of kinetic parameters until it provides a satisfactory description of the performance of the containment unit.

Detailed characterisation

The results of the first simulations, calibration and a sensitivity analysis can be used to define an additional (more thorough) sampling campaign with a more detailed influent characterisation (in relevant points along the system), and lab-scale tests for the determination of the key modelling parameters. The needs and characteristics of such a detailed sampling campaign can also be defined based on the uncertainty analysis.

Validation

The calibrated model needs to be validated by assessing its capacity to predict the performance of the plant using operational and environmental data from a different period than that used for the model calibration. If it fails the validation step, the model will need to be re-calibrated iteratively until a satisfactory validation is reached.

Study

A validated model can then be used to assess the scenarios of concern in accordance with the purpose and objectives of the modelling study.

Because of its practical nature and satisfactory application for model wastewater treatment plants, the steps of the STOWA calibration protocol will be reviewed from a faecal sludge modelling perspective, suggesting how they could be extrapolated and adapted to the particular characteristics and features of the most common onsite sanitation systems. This will be used to suggest the required steps towards developing a framework to model sanitation systems whose aim is to describe the dominant processes that take place inside the sanitation systems, in order to estimate the volumes and characteristics of the faecal sludge generated. However, one should bear in mind that while this framework describes different considerations and assumptions that need to be followed, but that also need to be proven and validated by applying and testing the framework and its outcomes in different sanitation systems. Ultimately, a structured and continuous application of the framework could lead over the years to a robust and solid protocol that could be applied with confidence and reliability, as has been observed in the wastewater field (Henze *et al*., 2008; Van Loosdrecht *et al*., 2016).

6.3 TOWARDS AN ONSITE SANITATION MODELLING FRAMEWORK

6.3.1 Onsite sanitation modelling: formulation of objectives

The first step is to define the main objectives of carrying out an onsite sanitation modelling study. Considering the prime purpose of sanitation, the main initial objectives should focus on (*i*) providing a tool to describe the accumulation of solids in onsite containment and treatment systems (as a function of the feeding rates and sludge disintegration) and to assess potential strategies to minimise the volumes of sludge, (*ii*) studying pathogen inactivation mechanisms, and evaluating different approaches to enhance and maximise the inactivation of pathogens, (*iii*) improving the prediction of the characteristics of the sludge contained, accumulated and emptied (as a function of the operating and environmental conditions of the sanitation systems) as a tool to contribute to improving the decision-making process in the sanitation chain, and (*iv*) evaluating the potential recovery of resources by maximising biogas production and enhancing nutrient recovery.

Different modelling studies have already been conducted (*i*) to describe the accumulation of solids (Brouckaert *et al*., 2013; Todman *et al*., 2015; Lugali *et al*., 2016; Strande *et al*., 2018); (*ii*) to model pathogen inactivation by pH, temperature or high ammonia concentrations in containment and treatment sanitation systems (Lübken *et al*., 2007; Fidjeland *et al*., 2013; Koottatep *et al*., 2014; Magri *et al* ., 2015); (*iii*) to model the anaerobic degradation of faecal sludge with special emphasis on biogas production (Elmitwalli *et al*., 2006, 2013; Wendland, 2008); and (*iv*) to study the aerobic degradation of faecal sludge (Lopez-Zavala *et al*., 2004a, 2004b). Most of these studies were conducted following empirical approaches and black-box models to achieve a satisfactory description of the accumulation of solids (Brouckaert *et al*., 2013; Todman *et al*., 2015; Lugali *et al*., 2016; Strande *et al*., 2018).

However, to include and consider additional and intermediate (biological and chemical) conversion processes could provide additional advantages that improve the operation of such systems. For instance, the hydrolysis and fermentation processes involved in the degradation of organic matter are often neglected, but these processes and their by-products can have an important influence on pathogen inactivation (Fidjeland *et al*., 2013; Magri *et al*., 2015; Anderson *et al*., 2015). There are also other models available and applied to describe the degradation of faecal sludge in lab-scale systems operated under well controlled conditions to forecast degradation efficiencies and performance (Lopez-Zavala *et al*., 2004a, 2004b; Wendland, 2008; Elmitwalli *et al*., 2006, 2013). These models need to be validated under actual operating and environmental conditions with real data measurements.

Last but not least, the pathogen inactivation models available so far tend to be stand-alone expressions (Lübken *et al*., 2007; Fidjeland *et al*., 2013; Koottatep *et al*., 2014; Magri *et al*., 2015) that need to be incorporated into mechanistic faecal sludge conversion and degradation models in order to explore different practical alternatives to enhance pathogen inactivation. Overall, the information and knowledge generated and provided by existing models are very valuable and can be combined and used to propose a basis to develop an expanded and structured mechanistic (glass-box) model for onsite containment and treatment sanitation systems that can be used to achieve the aforementioned objectives.

6.3.2 Onsite sanitation modelling: process description

There is a need to conceptually describe the activities and processes that take place in onsite containment and treatment systems. In this regard, onsite containment and treatment units can range from portable toilets (only used for containment prior to emptying, transportation and treatment) to borehole and pit latrines, septic tanks, and anaerobic baffled reactors. In order to define potential modelling approaches that reasonably represent the broader range of onsite sanitation systems, three commonly used technologies will be assessed in detail in this chapter: a portable toilet, a single pit latrine, and a septic tank. Because of the large variations in nature,

other onsite containment sanitation systems require different modelling approaches which fall outside the scope of this chapter. Nevertheless, for the sake of completeness, an overview of different models that can be applied to describe different onsite sanitation systems is presented later in this chapter (Table 6.3).

6.3.2.1 Portable toilets

A portable toilet is placed in a defined location to provide a temporary service that can range from a few days to months, and sometimes much longer, for example under emergency situations (Brdjanovic *et al.*, 2015). It is usually made of light, yet durable, material (plastic, PVC, wood, among others) to facilitate its transportation and has no large compartments to store high volumes of solids or liquids. It may have separated compartments (urine diversion toilet - UDT) to collect urine and faeces. There are three types: dry, pour flush, and flush. The latest generation can have three compartments, one each for urine, faeces and grey water, and can even include an extra compartment for internal storage for service water (as a source of grey water). An example of such a toilet, which was recently used in peri-urban areas of Nairobi, is shown in Figure 6.3.

Figure 6.3 Portable eSOS Smart Toilet with storage for urine, faeces and grey water, and a service water reservoir as the roof (image: Flex/design).

Since it can be used frequently (*e.g.* up to 300-400 times a day during public events or under emergency situations), these containment units fill up rapidly and require emptying. Some may need to be emptied every day whereas other toilets with larger storage volumes may operate for up to 7-10 days without being emptied (Zakaria *et al.*, 2017). When several single toilets are clustered (*e.g.* four or more), it is common to find larger containers, which makes the emptying periods less frequent. Taking into account that portable toilets do not have large compartments and that (consequently) they are emptied frequently, the faecal sludge and urine contained are usually fresh and of high strength (Lopez-Vazquez *et al.*, 2013; Zakaria *et al.*, 2018). Moreover, if the containment units are dry toilets made of impermeable materials (such as plastic) and often located above the ground (raised latrines), they are not subject to infiltration or seepage. Thus, the only input is the filling rate at which they are subject to by the users and the only output is due to emptying. The relatively high filling rates and emptying frequencies that result in short retention times allow little anaerobic or aerobic degradation of the faecal sludge. Arguably, this unit resembles a 'fill and draw' batch type of system (Henze *et al.*, 2007) (Figure 6.4).

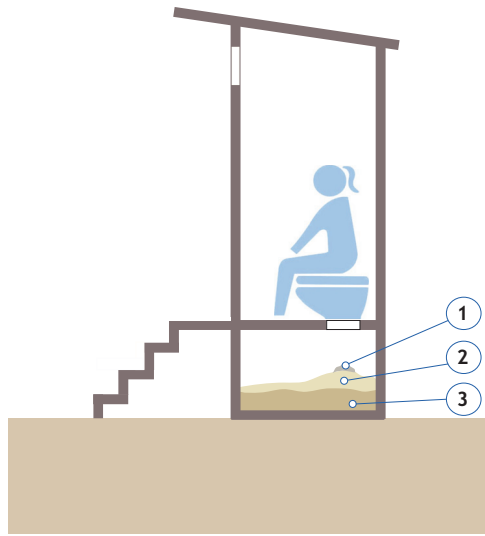


Figure 6.4 Schematic representation of a portable toilet without separated collection of urine (adapted from Bakare *et al.*, 2012).

In this system (and following the approach defined by Bakare *et al*. (2012)), three zones can be identified (Figure 6.4):

[1] Zone 1: the upper part where fresh faecal sludge and urine accumulate and are distributed over the cross-sectional area of the system.
[2] Zone 2: where the fresh faecal sludge and urine are already distributed. They remain in contact with the atmosphere, creating (micro-)aerobic conditions. In this zone the biological and chemical conversion processes start to take place.
[3] Zone 3: due to the accumulation of faecal sludge and urine and the consumption of oxygen in zone 2 (where the biological conversions under micro-aerophilic conditions start), zone 3 starts where the dissolved oxygen can no longer penetrate. As such, zone 3 is anaerobic and it triggers the occurrence of anaerobic conversion processes.

In zone 1, fresh faecal sludge accumulates depending upon the feeding rates in accordance with the number of users (Brouckaert *et al*., 2013; Todman *et al*., 2015; Lugali *et al*., 2016; Strande *et al*., 2018). This fresh faecal sludge from zone 1 is probably exposed to micro-aerophilic conditions in the exterior and possibly anaerobic in the interior. However, any biological or chemically-induced activity will only be driven by the microorganisms present in the fresh faecal sludge itself and, consequently, the biological conversions (if any) may be negligible. Overall, in zone 1, it can be assumed that the characteristics of fresh faecal sludge and urine will remain practically unchanged. Then, these components will only be distributed over the cross-sectional area of the unit as a function of the rheology of the sludge.

Zone 2 starts where the biological and chemical processes also start. Chemical conversions may begin (such as the hydrolysis of urine, depending on the presence of urease) (Rubio-Rincón *et al*., 2014), which are affected by the quality of the water used for toilet flushing, anal cleansing or washing the toilet. The fast filling rates (Zakaria *et al*., 2017), the high COD content of the faecal sludge (Strande *et al*., 2014; Chapter 2), and a potentially minimal diffusion of oxygen due to the physical characteristics of the faecal sludge (and merely driven by the atmospheric pressure) (Allaire *et al*., 2008) probably limit the availability of dissolved oxygen down to only a few millimetres in the solids layers. This suggests that zone 2 may be only a thin micro-aerophilic layer of a few millimetres that goes from the exterior layer up to where dissolved oxygen penetrates. Due to the limited availability and diffusion of oxygen, only some of the aerobic hydrolysis processes take place in zone 2 and a full aerobic conversion of organics cannot be expected. This is also because the relatively short retention times (as a consequence of the extremely frequent filling and emptying rates) will limit the accumulation of sludge and organisms.

Zone 3 starts where dissolved oxygen is no longer detected. Therefore, zone 3 is anaerobic and will trigger the anaerobic conversion of compounds. However, the short retention times (unless certain biomass/sludge is unwantedly retained after emptying inside the containment unit) will limit the growth of anaerobic bacteria (in particular the growth of methanogens) (Jabłoński *et al.*, 2015) suggesting that hydrolysis, some fermentation and (as much) a marginal acetogenesis process may be the dominant (biological) mechanisms. Thus, a full anaerobic conversion of the organics may not be expected. Therefore, inert or unbiodegradable compounds will not be excessively generated and accumulated in these systems unless the retention time is extended for some weeks or months. In zone 3 chemical processes are also expected to take place after the hydrolysis of urine and of other organic compounds has occured. This, in combination with the particular quality of used water or the addition of external compounds (*e.g.* magnesium or iron salts) (Zhang *et al*., 2008), may lead to the formation of certain crystals (*e.g.* calcium phosphate, and struvit) (Udert *et al*., (2003).

6.3.2.2 Single pit latrines

Another widely used onsite sanitation system is the single pit latrine. Excreta, along with anal cleansing materials or water, are deposited into the pit. They are emptied with a frequency that ranges from a few months (4-6 months) or a few years (1-2 years) to several years (even longer than 10 years) depending on the faecal sludge accumulation rate, which is the

function of percolation, degradation and consolidation of collected sludge (Broukaert *et al.,* 2013*;* Todman *et al*., 2015; Zziwa *et al.*, 2016). In one of the first efforts made to describe the accumulation of faecal sludge in these systems in more detail, Bakare *et al*. (2012) provided a conceptual description of the main processes that take place in a pit latrine.

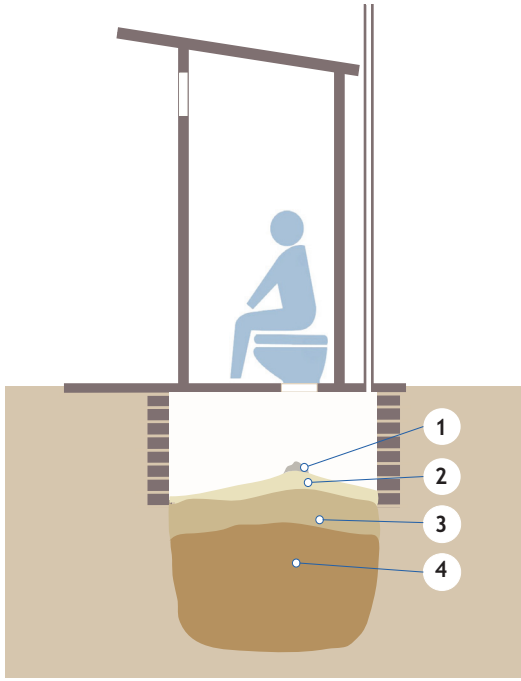


Figure 6.5 Schematic diagram illustrating the different theoretical layers within a pit latrine (adapted from Bakare *et al.*, 2012).

They identified four zones (Figure 6.5):

[1] Zone 1: the upper part of where fresh faecal sludge and urine will only accumulate and be distributed over the cross-sectional area of the system.

[2] Zone 2: in this zone the fresh faecal sludge and urine are already distributed in the system. They remain in contact with the atmosphere, creating (micro-) aerophilic conditions where the biological and chemical conversion processes start.

[3] Zone 3: due to the accumulation of faecal sludge and the consumption of oxygen in zone 2, the third zone starts when the dissolved oxygen can no longer penetrate, creating anaerobic conditions and therefore triggering the occurrence of anaerobic conversion processes.

[4] Zone 4: in the fourth zone, located at the bottom of the faecal sludge system, biological activity is minimal or no longer observed and only non-degradable or inert compounds accumulate.

A latrine is a larger, permanent system, which fills up and gets emptied less frequently then a portable toilet. Thus, the retention times are longer. This allows: (*i*) the retention of biomass, (*ii*) aerobic but mostly anaerobic conversions that can lead to the removal of organics and the accumulation of inert and non-degradable components, (*iii*) a substantial generation of gases (such as methane, carbon dioxide and also hydrogen sulphide), (*iv*) the occurrence of chemical processes and, (*v*) the infiltration of groundwater and the percolation/leakage of soluble components into the ground if the pit latrine is not well lined (sealed). Thus, zones 1 and 2 will be similar to those found in a portable toilet, zone 3 will allow the full anaerobic conversion of organics, and zone 4 will appear where most of the inert and non-degradable products from the conversion processes will accumulate. However, since it is a system with underground storage which is often an unlined pit, it is subject to the influence of the groundwater level, a particular problem in flood-prone areas. As such, it may suffer from groundwater infiltration that not only affects the biological and chemical conversions (*e.g.* due to the dilution effect as well as an increasing generation of gases if, for instance, sulphate-rich water intrudes into the latrine) but also allows the percolation of water and soluble compounds from the pit latrine into the ground.

The rheology of faecal sludge (Forster, 2002; Woolley *et al*., 2014a, 2014b; Liu *et al*., 2016) in combination with the impact of the infiltration of percolation processes will determine the way the faeces, urine and water are distributed and percolate between the different zones and will also affect the consumption and production of soluble and particulate products of the dominant conversion process(es) prevailing in each phase. The longer

retention times will allow an extended conversion of organic matter in zone 3 that will lead to the generation of gases (such as methane, carbon dioxide and possibly also hydrogen sulphide) that will mostly diffuse into zones 2 and 1. The diffusion of such gases into zone 2, in combination with the high organic loads present in the faecal sludge discharged into the latrine, will decrease the volume and thickness in zone 2. While the start of zone 3 can be determined based on the profile of dissolved oxygen, its depth and thickness cannot be easily determined. This is mostly because, as pointed out by Nwaneri *et al.* (2008), this phase finishes at a depth where the accumulation of inert and non-degradable compounds is dominant, meaning that zone 3 finishes where the anaerobic biological conversions become negligible or are no longer observed. Zone 4 starts where zone 3 finishes and in this layer mostly unbiodegradable or non-degradable organic and inorganic compounds accumulate.

6.3.2.3 Septic tanks

Septic tanks are a common onsite sanitation system. They can be relatively simple and made of concrete, fibreglass, vynil or plastic. They are composed of at least two compartments divided by one baffle (Figure 6.6). Excreta and anal cleansing materials are deposited into the septic tank. They are emptied with a frequency that ranges from a few (1-2 years) to several years (even longer than 10 years) (Broukhaert *et al*., 2013; Todman *et al*., 2015; Zziwa *et al*., 2016) depending on faecal sludge accumulation, but the hydraulic retention time can be as short as a few hours (12-24 hours) especially when the tank is full. Settleable solids accumulate at the bottom of the system whereas floating material accumulates at the top. Mostly anaerobic conversion processes contribute to the removal and reduction of the organic matter. From a process description perspective, the two (or more) compartments can be divided into different zones (Figure 6.6) as explained below.

Compartment 1:

[1] Zone 1: the upper part where the wastewater is received and settleable and non-settleable matter is split.

[2] Zone 2: a small (micro-)aerobic zone where some dissolved oxygen may be present, either from the influent or due to oxygen diffusion. Thus, some aerobic processes may take place.

[3] Zone 3: the anaerobic zone. This zone can be further divided into two sub-zones where the soluble compounds (3a) and the particulate compounds (3b) can be degraded separately, respectively.

[4] Zone 4: located at the bottom of the septic tank where the biological activity is minimal or no longer observed and only non-degradable or inert compounds accumulate.

Compartment 2:

[1] Zones 1 and 2: they cannot be found in the 2nd compartment since wastewater is already mixed and the dissolved oxygen concentrations are negligible.

[2] Zone 3: the anaerobic zone. Here anaerobic conversion processes of soluble and particulate organic matter (that do not settle in the 1st compartment) and residual reaction products produced in the 1st compartment take place.

[3] Zone 4: in this last zone, only non-degradable or inert compounds accumulate.

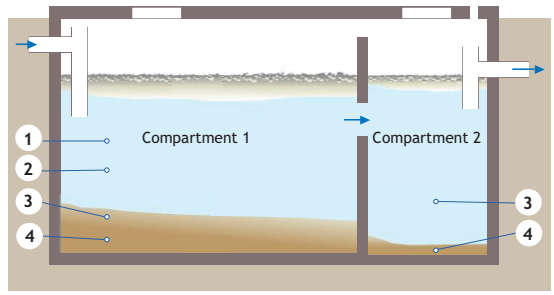


Figure 6.6 Schematic diagram illustrating the different theoretical layers within a septic tank (adapted from Tilley *et al.*, 2014).

The longer retention times of septic tanks and their configuration composed of two compartments divided by a baffle allows in the 1st compartment the development of four zones similar to those discussed previously for pit latrines. However, most of the settleable solids present in the influent wastewater settle in the 1st compartment and the rest flows to the

2nd compartment. The settleability of the solids influences the fraction retained in the 1st compartment and the fraction that flows to the 2nd compartment. Thus, in zone 3 of the 1st compartment, the fraction of the settleable solids retained in the 1st compartment degrades anaerobically and zone 4 accumulates the inert and non-degradable matter from the upper zones. It is possible that zone 3 needs to differentiate between the anaerobic degradation of soluble matter and suspended matter by splitting the zones in two. The gases generated from zone 2 and mostly from zone 3 diffuse into the adjacent zones.

The 2nd compartment is only composed of one zone 3 and one zone 4. In zone 3, the degradable matter not retained in the 1st compartment and the products and residual concentrations generated in zone 3 of the 1st compartment degrade anaerobically. Zone 4 of the 2nd compartment accumulates the remaining inert and non-degradable matter from zone 3. The wastewater flows out of the system from zone 3 of the 2nd compartment, determining the quality of the treated effluent. The gases generated in zone 3 diffuse into the headspace of the septic tank, into zone 4 and also leave through the effluent.

Being an underground system, similar to the pit latrine, septic tanks may be affected by groundwater infiltration influencing the biological and chemical conversions, as previously discussed, and also allowing the percolation of water and soluble compounds from each zone into the ground.

Overall, the portable toilet, the pit latrine and the septic tank have different conversion processes influenced by their configurations, use of water and type of service provision, location, and operation and maintenance. Tables 6.3a and 6.3b aim to provide a general overview of (*i*) the main conversion processes and (*ii*) the main transport processes that take place in these systems.

Table 6.3a General overview of the main conversion processes in portable toilets, pit latrines and septic tanks.

	Portable toilet	Pit latrine	Septic tank	
			1st compartment	2nd compartment
Retention time	Short - usually less than a few days (*e.g.* 7 days).	Long - varying from a few to several years (1-20 years).	Long - varying from a few to several years (1-20 years).	
Main aerobic conversion processes (zone 2).	(Micro-) aerobic zone of a few mm defined by the penetration of dissolved oxygen. Aerobic hydrolysis takes place but full aerobic conversion of organics is not expected.	Aerobic hydrolysis and (marginal) heterotrophic organic matter removal.	Aerobic hydrolysis and (marginal) heterotrophic removal of soluble organic matter (limited by oxygen diffusion and availability).	None. Absent. Full anaerobic compartment.
Main anaerobic conversion processes (zone 3).	Hydrolysis and fermentation.	Hydrolysis, fermentation, acetogenesis and methanogenesis	Hydrolysis, fermentation, acetogenesis and methanogenesis (mostly on settled compounds).	Hydrolysis, fermentation, acetogenesis and methanogenesis (mostly on soluble compounds).
Accumulation of inert and non-degradable matter (zone 4).	No accumulation, due to short retention times (zone 4 does not exist).	Accumulation in zone 4, due to long retention times.	Accumulation in zone 4, due to long retention times. Mostly produced by the anaerobic conversions of particulate compounds retained in 1st compartment.	Accumulation in zone 4, due to long retention times.

Table 6.3b (Part 1 of 2) General overview of the main transport mechanisms in portable toilets, pit latrines and septic tanks.

	Portable toilet	Pit latrine	Septic tank	
			1st compartment	2nd compartment
Zone 1	*Inputs*			
	• Faecal sludge and urine as function of filling rates.	• Faecal sludge and urine as function of filling rates. • Groundwater infiltration.	• Faecal sludge and urine as function of filling rates. • Diffusion of dissolved oxygen from atmosphere and of gases from zone 2. • Groundwater infiltration.	
	Outputs			
	• Percolation of faecal sludge and urine to zone 2.	• Percolation of faecal sludge and urine to zone 2.	• Soluble compounds flow to 2nd compartment and also diffuse into zone 2. • A large fraction of particulate or suspended matter settles and reaches zone 2, the remaining fraction flows to the 2nd compartment. [Zone 1 not well defined.]	[Zone 1 is absent.]
Zone 2	*Inputs*			
	• Soluble and particulate compounds of faecal sludge and urine from zone 1. • Diffusion of dissolved oxygen from atmosphere.	• Soluble and particulate compounds of faecal sludge and urine from zone 1. • Diffusion of dissolved oxygen from atmosphere and of gases from zone 3 (*e.g.* methane, carbon dioxide, hydrogen sulphide). • Groundwater infiltration.	• Soluble and particulate compounds of faecal sludge and urine retained in 1st compartment. • Diffusion of dissolved oxygen from atmosphere and of gases from zone 3 (*e.g.* methane, carbon dioxide, hydrogen sulphide). • Groundwater infiltration.	
	Outputs			
	• Percolation of aerobically converted products of faecal sludge and urine to zone 3.	• Percolation of aerobically converted products of faecal sludge and urine and inert or non-degradable compounds to zone 3. • Infiltration into the ground.	• Percolation of aerobically converted products of faecal sludge and urine and inert or non-degradable compounds to zone 3. • Infiltration into the ground.	[Zone 2 is absent.] [No oxygen is available.]

Table 6.3b (Part 2 of 2) General overview of the main transport mechanisms in portable toilets, pit latrines and septic tanks.

	Portable toilet	Pit latrine	Septic tank	
			1st compartment	2nd compartment
Zone 3	*Inputs*			
	• Percolation of aerobically converted products of faecal sludge and urine from zone 2.	• Products of faecal sludge and urine from zone 2. • Inert or non-degradable compounds from zone 2. • Groundwater infiltration.	• Products of faecal sludge and urine from zone 2. • Inert or non-degradable compounds from zone 2. • Groundwater infiltration.	• Mostly soluble and the fraction of the particulate compounds of faecal sludge and urine not retained in 1st compartment. • Soluble products of faecal sludge and urine from zone 3 of 1st compartment. • Diffusion of gases generated in the zone 3 of 1st compartment (*e.g.* methane, carbon dioxide, hydrogen sulphide). • Inert or non-degradable soluble compounds from zone 3 of 1st compartment. • Groundwater infiltration.
	Outputs			
	• No outputs.	• Percolation of anaerobically degraded inert and non-degradable matter to zone 4. • Diffusion of gases generated to zones 2 and 4 (*e.g.* methane, carbon dioxide, and hydrogen sulphide). • Infiltration into the ground.	• Percolation of anaerobically degraded inert and non-degradable matter to zone 4 of 1st compartment. • Diffusion of gases generated to zones 2 and 4 (*e.g.* methane, carbon dioxide, and hydrogen sulphide). • Infiltration into the ground.	• Effluent. • Percolation of anaerobically degraded inert and non-degradable matter to zone 4 of 2nd compartment. • Diffusion of gases generated to zone 4 (*e.g.* methane, carbon dioxide, hydrogen sulphide).
Zone 4	*Inputs*			
		• Accumulation of percolated anaerobically degraded compounds and/or inert and non-degradable matter from zone 3. • Groundwater infiltration.	• Accumulation of percolated anaerobically degraded compounds and/or inert and non-degradable matter from zone 3. • Groundwater infiltration.	• Accumulation of percolated anaerobically degraded compounds and/or inert and non-degradable matter from zone 3 of 2nd compartment. • Groundwater infiltration.
	Outputs			
	[Zone 4 is absent.]	• Desludging. • Infiltration into the ground.	• Desludging. • Infiltration into the ground.	• Desludging. • Infiltration into the ground.

6.3.3 Onsite sanitation modelling: data collection and verification

Data need to be collected for five main purposes: (*i*) to determine the volumes of faecal sludge including urine, (*ii*) to determine the characteristics of faecal sludge and urine, (*iii*) to define the length of the reaction zones in each system, (*iv*) to assess the conversion processes, and (*v*) to estimate possible infiltrations and percolation flows. Table 6.4 suggests different sampling campaigns to assess the first four of these purposes.

Most of the samples collected for the analytical determination of standard parameters can follow the corresponding recommendations for sampling, preservation, transportation and storage (Chapter 3) prior to the conduction of the analytical tests. However, for the conduction of the required (anaerobic and aerobic) biological, physical and chemical tests, it is important to collect reliable and representative samples from each layer at different depths that have not been adulterated or disturbed. Sampling procedures from soil mechanics or studies in sediments need to be applied and followed to collect the required soil and sludge samples and transport and store them prior to the conduction of the batch activity tests of interest (Strande *et al*., 2014).

Due to public health concerns, it is essential to assess the transport and distribution of pathogens, viruses and other harmful bacteria or organisms between the different zones. In parallel, the formation and accumulation of products, compounds and elements from the biological and chemical processes may influence the viability and inactivation of pathogens, viruses and other harmful bacteria or organisms. Several studies have been carried out on the transport of pathogens in faecal sludge and porous media (Mensah *et al*., 2013) and these could be used to execute the required tests with samples of solids collected at different depths, and linked with the assessment and effects of potential inhibitory elements or compounds from the biological and chemical conversions at the different layers.

In the case of septic tanks, it is recommended to also conduct tracer tests for a better determination of the hydrodynamic behaviour of the system and to define the hydraulic residence time (Metcalf and Eddy *et al*., 2014).

Infiltration and percolation mechanisms affect the transport of the components and elements of interest, also influencing the conversion processes in each zone and the performance of the system as a whole. These mechanisms and their rates are not only dependent on the rheology of the solids or characteristics of the system but also on hydrological and groundwater processes (Foppen, 2002; Halalsheh *et al*., 2011). All these processes need to be studied in a structured manner and would probably, as with many other processes, be case-specific for each location.

The quality and reliability of the measurements need to be verified through the conduction of mass balances on water, COD, nitrogen and phosphorus. However, in addition to these balances, when reviewing the performance of anaerobic systems, molar balances also need to be performed because carbon dioxide (not accounted for in the COD balance) will be generated which affects the composition of the biogas produced, the pH and even the ADM1 model stoichiometry (Klerebezeem and Van Loosdrecht, 2006a, 2006b; Rodriguez *et al*., 2006). Another reason for performing molar balances is that they are different anaerobic processes that are pH-dependent. If there are major differences to close the mass balances (higher than 10-15%), it will be necessary to check the results of the analytical parameters and thereafter the configuration of the system to conduct another (detailed) sampling campaign.

Table 6.4 (Part 1 of 2) Suggested sampling campaigns for data collection and verification for modelling of a portable toilet, pit latrine and septic tank.

	Portable toilet	Pit latrine	Septic tank	References
Purpose	*Determination of volumes of faecal sludge, urine or wastewater*			
Duration	1-2 days	2-3 days	2-3 days	See chapters 2, 3 and 5.
Frequency	Continuous recording of no. of users during representative periods of use.	Continuous recording of no. of users during representative periods of use.	Assessment over a few hours	
Purpose	*Determination of characteristics of faecal sludge and urine*			
Duration	1-2 days	2-3 days	2-3 days	See chapters 2, 3 and 5. Mensah *et al*. (2013)
Frequency	Every 2-3 hours	Every 2-3 hours	Every 2-3 hours	
Type of samples	Composite	Composite	Composite	
Parameters	Total COD, soluble COD, TSS, VSS, TN, NH_4-N, TP, PO_4-P, pH, microbiological analyses, sludge rheology, and dewaterability.	Total COD, soluble COD, TSS, VSS, TN, NH_4-N, TP, PO_4-P, pH, microbiological analyses, sludge rheology, and dewaterability.	Total COD, soluble COD, TSS, VSS, TN, NH_4-N, TP, PO_4-P, pH, microbiological analyses, settleable matter, and floating matter.	
Purpose	*Determination of the length of reaction zones*			
Duration	1-2 days	1-2 days	1-2 days	See Chapter 3.
Frequency	Every 2-3 hours	Every 2-3 hours	Every 2-3 hours	
Type of samples	Use of portable meters. Collection of undisturbed solids samples to perform aerobic and anaerobic activity tests.	Use of portable meters. Collection of undisturbed solids samples to perform aerobic and anaerobic activity tests.	Use of portable meters. Collection of undisturbed solids samples to perform aerobic and anaerobic activity tests.	
Sampling locations	Vertical and horizontal directions across the system.	Vertical and horizontal directions across the system.	Vertical and horizontal directions across the system in each compartment.	
Parameters	• DO (if available, use a microelectrode) and redox potential. • Conduction of experimental methods to assess aerobic and anaerobic activities.	• DO (if available, use a microelectrode) and redox potential. • Conduction of experimental methods to assess aerobic and anaerobic activities.	• DO (if available, use a microelectrode) and redox potential. • Conduction of experimental methods to assess aerobic and anaerobic activities.	

Table 6.4 (Part 2 of 2) Suggested sampling campaigns for data collection and verification for modelling of a portable toilet, pit latrine and septic tank.

	Portable toilet	Pit latrine	Septic tank	References
Purpose	*Assessment of conversion processes*			
Duration	1-2 days	1-2 days	1-2 days	See chapters 2, 3 and 5. Van Loosdrecht *et al*. (2016).
Frequency	• Every 3-4 hours (grab samples) • 24 hours (composite samples)	• Every 3-4 hours (grab samples) • 24 hours (composite samples)	• Every 3-4 h (grab samples) • 24 hours (composite samples)	
Type of samples	• Grab • Composite	• Grab • Composite	• Grab • Composite	
Sampling locations	• Grab and composite samples in each reaction zone.	• Grab samples in the influent and effluent and in the interface between reaction zones. • Composite samples in each zone.	• Grab samples in the influent and effluent and in the interface between reaction zones in each compartment. • Composite samples in each zone.	
Parameters	• Grab samples: - Total COD, soluble COD, TSS, VSS, TN, NH_4-N, TP, PO_4-P, pH, microbiological analyses, sludge rheology, and dewaterability.	• Grab samples: - Total COD, soluble COD, TSS, VSS, TN, NH4-N, TP, PO_4-P, pH, microbiological analyses, sludge rheology, and dewaterability.	• Grab samples: - Total COD, soluble COD, TSS, VSS, TN, NH_4-N, TP, PO_4-P, pH, microbiological analyses, settleable matter, and floating matter.	
Tests and methods	• Composite samples: - Aerobic and anaerobic activities. - Aerobic and anaerobic fractionations and biodegradation rates.	• Grab samples: - Off-gases (methane, carbon dioxide, hydrogen sulphide). • Composite samples: - Aerobic and anaerobic activities. Aerobic and anaerobic fractionations and biodegradation rates. - Chemical precipitation tests.	• Grab samples: - Off-gases (methane, carbon dioxide, hydrogen sulphide). • Composite samples: - Aerobic and anaerobic activities. Aerobic and anaerobic fractionations and biodegradation rates. - Chemical precipitation tests.	

6.3.4 Onsite sanitation modelling: model structure

6.3.4.1 Model structure of commonly used onsite sanitation systems

In this section, three representative and commonly used faecal sludge containment and/or treatment systems are assessed in detail with the aim of defining basic structures and highlighting the required information and assumptions that need to be gathered and proposed to model these systems. The containment systems selected and subject to a deeper discussion and assessment are the portabletoilet, the pit latrine and the septic tank. This approach is also based on the consideration that more complex systems, such as the anaerobic-baffled reactors or the upflow anerobic sludge blanket (UASB) reactors, could probably be developed based on the basic structures suggested for these three more basic units but expanded (both 'physically' by considering a higher number of interconnected reactors and also with regard to the process performance by incorporating more complex models). Therefore, in the next section, after the discussion of these three basic units, some suggestions are given to model more complex onsite containment and treatment systems (Section 6.3.4.2).

Portable toilets

The first model structure suggested is for a portable toilet (Figure 6.7). As discussed in Section 6.3.2.1, these are usually a closed system with a short retention time (of maximum a few weeks), it is composed of three zones or phases: zone 1 where the sludge retains its physical properties, zone 2 where it is distributed and contains dissolved oxygen that drives certain aerobic conversions, and zone 3 where the conditions become anaerobic and anaerobic conversions take place. In this suggested model structure, it is assumed that the relatively short retention time (of a few weeks) does not allow the complete conversion and degradation of the organics. Consequently, only a marginal degradation or conversion of the degradable matter is reached. There is no gas generation (since the conversions are not complete) and zone 4 is absent. When present, the function of zone 4 is to retain and accumulate the inert and non-degradable matter present in the influent or produced from the degradation processes. The fluxes of soluble (S) and suspended (X) compounds are indicated (Q_{1_2} and Q_{2_3}, for their transport from zone 1 to zone 2, $S_{FS,1\_2}$ and $X_{FS,1\_2}$, and from zone 2 to zone 3, $S_{FS,2\_3}$ and $X_{FS,2\_3}$, respectively) including the presence and transport of pathogens between zones ($X_{pathogens,inf}$, $X_{pathogens,1\_2}$, $X_{pathogens,2\_3}$). The system is fully closed and the only input is the discharge of faecal sludge, urine and water and the only output is the periodic emptying rate ($Q_{FS,emptying}$), resembling a fill-and-draw system. This can be considered the simplest model structure for a faecal sludge system.

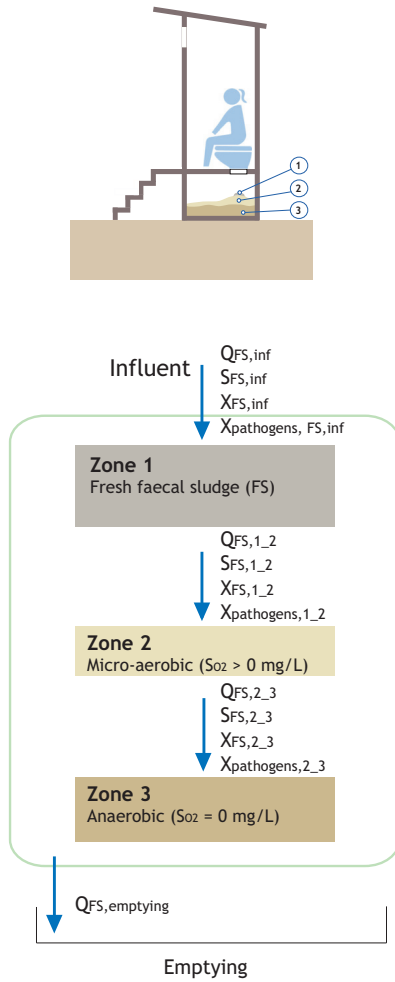


Figure 6.7 Portable toilet: suggested model structure.

Pit latrines

Pit latrines are more complex than portable toilets (Figure 6.8). Although they are subject to some similar conditions, they have longer retention times (of several months and even years) that result in the full completion of the conversion processes (mostly the anaerobic ones).

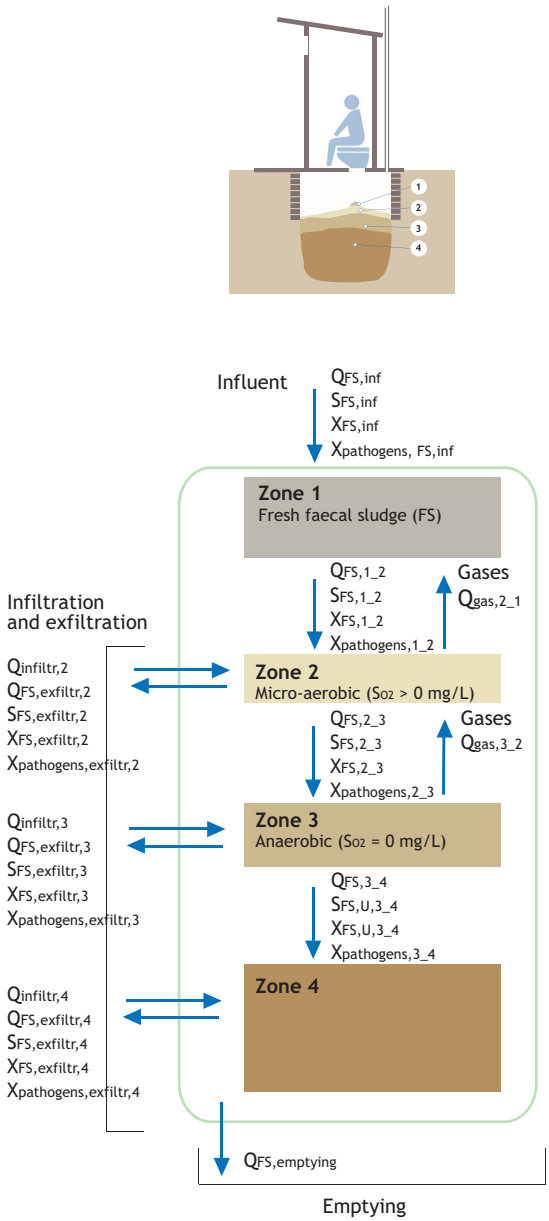
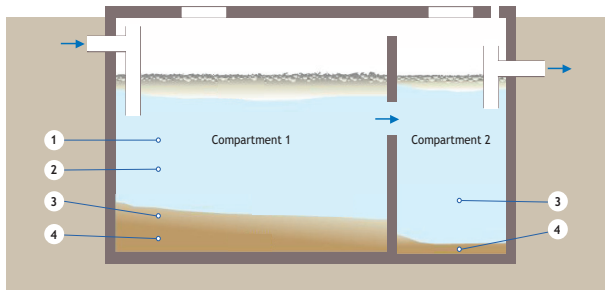


Figure 6.8 Pit latrine: suggested model structure.

This implies that the kinetics will probably not play a major role and that stoichiometric relationships can be used to describe the conversion processes. This has already been observed in studies by Brouckaert *et al*. (2013) and Todman *et al*. (2015) who were able to model the filling rates of pit latrines using basic kinetic expressions. Moreover, pit latrines are prone to infiltration and percolation. Thus, besides the effects of the transport phenomena of the sludge matrix and associated processes between zones (*e.g.* $Q_{FS,1\_2}$ that transports the soluble, $S_{FS,1\_2}$, and particulate concentrations, $X_{FS,1\_2}$ and $X_{pathogen,1\_2}$, from zone 1 to zone 2), pit latrines may also dilute their concentrations due to the infiltration of groundwater (*e.g.* $Q_{infiltr,2}$ for the infiltration in zone 2) and/or concentrate the particulate compounds because of the percolation rates (for instance, $Q_{exfiltr,2}$ to describe the exfiltration of compounds $S_{FS,exfiltr,2}$, $X_{FS,exfiltr,2}$ and $X_{pathogens,exfiltr,2}$ from zone 2). Gases and inert and non-degradable matter ($S_{FS,U}$ and $X_{FS,U}$) are usually generated, since the anaerobic conversion processes are completed. On the one hand, this leads to the transport and diffusion of gases between zones (*e.g.* $Q_{gas,2\_1}$ and $Q_{gas,3\_2}$ for the gas emissions from zone 2 to the atmosphere and from those of zone 3 to zone 2, respectively). On the other hand, due to inert and non-degradable products from the anaerobic processes remaining in zone 3, this leads to their transport from zone 3 to zone 4 ($S_{FS,U,3\_4}$, $X_{FS,U,3\_4}$) and accumulation at the bottom of the system leading to the creation of an inert zone (zone 4). Similar to the portable toilets, the model structure of the pit latrine has one major input (the sludge feed, $Q_{FS,inf}$) and one major output (the emptying rate, $Q_{FS,emptying}$), but also the infiltration ($Q_{infilt,2}$, $Q_{infilt,3}$, $Q_{infilt,4}$) and exfiltration rates ($Q_{FS,exfiltr,2}$, $Q_{FS,exfiltr,3}$, $Q_{FS,exfiltr,4}$) that may affect each zone to different degrees. These also affect the soil and groundwater quality (due to the exfiltration of the soluble and particulate compounds (*e.g.* the compounds $S_{FS,exfiltr,4}$, $X_{FS,exfiltr,4}$ and $X_{pathogen,exfiltr,4}$ flow from zone 4 into the ground).

Septic tanks

Compared to pit latrines, septic tanks usually receive a combination of faecal sludge and water (domestic wastewater) and are usually divided into two compartments (Figure 6.9). They work in a continuous mode and have long retention times (of years) that, similar to pit latrines, will result in full completion of the conversion processes (mostly the anaerobic ones). This implies that stoichiometric conversion ratios can be sufficient to provide a satisfactory description of the processes that take place in these units. Septic tanks are also prone to infiltration and percolation issues. Therefore, they have well defined inputs ($Q_{FS,inf}$, $Q_{WW,inf}$) and output (Q_{eff}) but are prone to infiltration and percolation flows. Practically all the settleable solids present in the input tend to be retained in the 1st compartment while non-settleable solids flow to the 2nd compartment ($S_{FS,inf}$, $X_{FS,1.1\_2.1}$ and $X_{pathogens,1.1\_2.1}$).



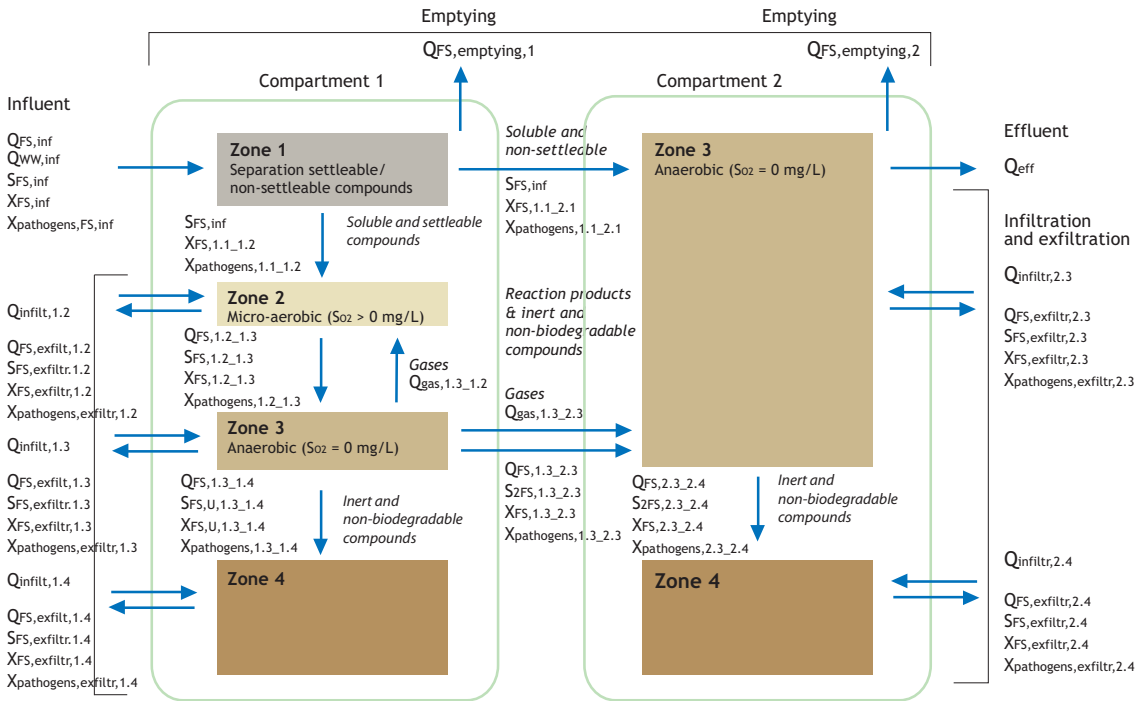


Figure 6.9 Septic tank: suggested model structure.

The settleable solids need to be measured to split the flows between the two compartments. The expected low oxygen diffusion in the 2nd compartment and the split in the flow lead to the existence of four zones in the 1st compartment (similar to those proposed for pit latrines) but only two in the 2nd compartment. In the 1st compartment, most of the processes take place in the settleable solids and soluble components and, in the 2nd compartment, in the non-settleable solids and soluble components. In addition, the 2nd compartment receives the reaction products from zone 3 of the first compartment. Consequently, a higher accumulation of solids can be expected in the 1st compartment ($S_{FS,U,1.3\_1.4}$ and $X_{FS,U,1.3\_1.4}$) than in the 2nd compartment ($S_{FS,U,2.3\_2.4}$ and $X_{FS,U,2.3\_2.4}$).

6.3.4.2 Model structures of other sanitation systems

There are several onsite and sewered sanitation technologies found in sanitation practice to which a similar approach and structure as proposed above can be applied (Table 6.5).

Table 6.5 (Part 1 of 2) Suggestions for faecal sludge modelling of sanitation technologies (list of technologies adopted from Tilley *et al*., 2014).

Technology	Suggested models	Confidence	Track record	Applicability	Suggested literature for further reading
Portable toilets	ASM+ADM1	Low	Limited	Low	Henze *et al*. (2000), Batstone *et al*. (2000), Lopez-Zavala *et al*. (2004a, 2004b), Elmitwalli *et al*. (2006, 2011, 2013).
Pit latrines (single ventilated improved pit, double ventilated improved pit)	ASM+ADM1	Low	Medium	Low	Henze *et al*. (2000), Batstone *et al.* (2000, 2015), Brouckaert *et al*. (2013), Lopez-Zavala *et al.* (2004a, 2004b), Elmitwalli *et al.* (2006, 2011, 2013).
Septic tank with multiple units	ASM+ADM1	Low	Medium	Low	Henze *et al.* (2000), Batstone *et al.* (2000, 2015), Lopez-Zavala *et al*. (2004a, 2004b), Elmitwalli *et al*. (2006, 2011, 2013).
Fossa alterna	ASM+ADM1	Low	Limited	Low	Henze *et al.* (2000), Batstone *et al.* (2000, 2015), Mata-Alvarez *et al*. (2011), Girault *et al*. (2012),
Twin pits for pour flush	ASM+ADM1	Low	Limited	Low	Henze *et al.* (2000), Batstone *et al.* (2000, 2015), Lopez-Zavala *et al*. (2004a, 2004b), Elmitwalli *et al.* (2006, 2011, 2013).
Anaerobic baffled reactor	ADM1	Medium	Limited	Low	Barber and Stuckey (1999), Batstone *et al.* (2000), Skiadas *et al* (2000), Zhu *et al*. (2015).
Anaerobic filter	ADM1+biofilm model	Medium	Limited	Medium	Batstone *et al*. (2000, 2015), Saravanan and Sreekrishnan (2006), Rittman *et al.* (2018)

Table 6.5 (Part 2 of 2) Suggestions for faecal sludge modelling of sanitation technologies.

Technology	Suggested models	Confidence	Track record	Applicability	Suggested literature for further reading
Imhoff tank	ADM1	Medium	Medium	Medium	Batstone *et al*. (2000, 2015), Donoso-Bravo *et al.* (2011,), Mata-Alvarez *et al*. (2011), Eltmitawili *et al.* (2001, 2011, 2013), Wendland (2009).
Waste stabilisation ponds	ADM1 + ASM + hydraulic models	Medium	Medium	High	Henze *et al*. (2000), Batstone *et al*. (2000), Shilton and Harrison (2003), Alvarado *et al.* (2012), Sah *et al.* (2012).
Aerated pond	ASM	High	Medium	High	Henze *et al.* (2000), Houweling *et al.* (2005, 2008), Alvarado *et al.* (2012), Sah *et al.* (2012).
Wetlands	ADM1	High	Extensive	High	Henze *et al.* (2000), Batstone *et al*. (2000), Langergraber *et al.* (2009), Bridgham *et al.* (2013).
Trickling filter	ASM+biofilm model	Medium	Medium	Medium	Henze *et al*. (2000), Wik (2003), Rittman *et al*. (2018).
Upflow anaerobic sludge blanket system	ADM1	High	Extensive	High	Batstone *et al*. (2000, 2005), Eltmitwalli *et al.* (2001, 2011, 2013), Saravanan and Sreekrishnan (2006), Wendland (2009), Donoso-Bravo *et al.* (2013).
Activated sludge	ASM + ADM1	High	Extensive	High	Henze *et al*. (2000), Lopez-Zavala *et al*. (2004a, 2004b), Lopez-Vazquez *et al*. (2013), Brdjanovic *et al*. (2015).

Providing a thorough modelling approach for each faecal sludge collection and treatment technology falls outside the scope of this chapter. However, because limited experience in faecal sludge modelling at this relatively early stage means that additional studies and data are required, a brief overview is provided that presents different potential models that can be used to model faecal sludge containment/treatment technologies and suggested literature for further reading that can be useful to (start to) develop the required models for these systems. In Table 6.5, the levels of confidence, track record and applicability refer to the reliability of the modelling experiences, the availability of studies and papers in the literature, and the number of case studies and full-scale applications of such models, respectively.

6.3.5 Onsite sanitation modelling: characterisation of flows

Prior to characterising the flows, it is important to define the sizes of the reaction zones. However, this is not a straightforward task because very often they do not have defined physical boundaries. After reviewing the process designs and model structure of the portable toilet, the pit latrine and the septic tank (figures 6.4 to 6.9), it is likely that most of the systems will be anaerobic since the diffusion of dissolved oxygen into the contents of these units will be very low. In soils, wetlands and in particular in peat soils (which may to some extent resemble faecal sludge sanitation systems), oxygen penetration is limited to the first ten centimeters (Ball *et al*., 1997; Armstrong *et al*., 2000) and sometimes to even the

first centimeter (Sexstone *et al*., 1985). Thus, methane production and consumption is observed within the first 10-20 cm just below the surface (Dunfield *et al.,* 1993). Moreover, the high organic concentrations observed in wastewater and faecal sludge (higher than 500-1,000 mgCOD/L) (Lopez-Vazquez *et al*., 2013) have a high oxygen demand. Also, the methane generated in the anaerobic zones of the sanitation systems (*e.g.* zone 3 in figures 6.4 to 6.6) may intrude into the aerobic zones and consume oxygen. As a consequence, if oxygen diffusion is not enhanced (*e.g.* by mixing or external aeration) (Stenstroom and Rosso, 2010), it is highly likely that the aerobic zone proposed for the previous sanitation systems (zone 2 in figures 6.4 to 6.6) will be very small (with a thickness of just a few millimetres) or even absent. To define the size of the reaction zones, it is proposed to conduct different measurements of dissolved oxygen and redox potential profiles both vertically and horizontally within the systems. If possible and since the aerobic zone may be very small, the use of microlectrodes (Revsbech and Jørgensen, 1986) is recommended to determine the size or, more specifically, the thickness of the aerobic zone, if any. To determine the size and volumes of the anaerobic and inert zones (zones 3 and 4 in figures 6.7 to 6.9), the collection of undisturbed samples or sludge cores at different heights can be helpful to conduct anaerobic batch activity tests (as well as to assess the microbial population dynamics and sludge characteristics). For this purpose, experience gathered in other fields (*e.g.* groundwater or paleolimnology) (Glew *et al*., 2002) can be very useful to guide the collection of undisturbed and representative sludge core samples at the required heights to carry out the required activity tests and analysis (see Chapter 3). The results of the execution of aerobic and anaerobic activity tests (Van Loosdrecht *et al*., 2016), combined with microbial identification studies (McIlroy *et al*., 2015) and the characterisation of the sludge, will provide valuable information to define the size and volume of each phase and reaction zone, whereas the inert zone will start at the height where activity is minimal or even ceases.

If most of the faecal sludge process conversions are aerobic, efforts can be made to describe the aerobic activity with the application of aerobic models (Lopez-Zavala *et al*., 2004a, 2004b). The rest of the conversion processes will be anaerobic (zone 3 in figures 6.4 to 6.6) or even the whole system will be anaerobic if there are no aerobic zones (as discussed previously). To model the anaerobic conversion processes, the most suitable model is IWA ADM1 (Batstone *et al*., 2002). Since it was launched, this anaerobic model has remained state-of-the-art and, with different extensions and modifications, been successfully applied to several anaerobic conditions and systems (Batstone *et al*., 2006, 2015; Donoso-Bravo *et al*., 2011; Kythreotou *et al*., 2014). Furthermore, ADM1 has already been adapted and applied to model the anaerobic treatment and degradation of faecal sludge, black water and household solid waste in onsite sanitation systems (Wendland, 2008; Elmitwalli *et al*., 2006, 2011, 2013). However, these models have not been calibrated or validated with actual measurements from real sanitation systems. They have been used as tools to foresee and explore potential process performance and process improvements for system selection either deriving input and operational data from previous studies or from lab-scale systems. This indicates that information and experience available to model real faecal sludge systems are still limited. Furthermore, there are key structural bottlenecks related to the required ADM1 fractionation and the fractionation of faecal sludge that need to be carefully addressed, as will be discussed in later sections of this chapter.

Once the zones are known, the flows between each zone can be characterised following the recommendations given in Section 6.3.3 on data collection and verification. However, in addition to the well-known solid-liquid and gas-liquid transport mechanisms, in faecal sludge systems it will also be necessary to assess the transport of pathogens and gases in porous media. While the solid-liquid and gas-liquid transport and diffusion phenomena can be assumed to be well understood and defined based on the knowledge gathered from wastewater treatment systems (Brdjanovic *et al*., 2015), the transport of pathogens through the different zones of faecal

sludge systems needs to be well-defined in order to understand and be able to describe their potential spatial distribution in faecal sludge systems. Previous reports describing the spatial distribution of pathogens in sanitation systems and past studies conducted on the transport of pathogens through porous media can be useful in this regard (Foppen *et al*., 2007a, 2007b, 2010). Once again, an appropriate collection of sludge cores (Glew *et al*., 2002) and the use of advanced molecular identification methods (Karst *et al*., 2016 can provide a useful overview to understand the physical distribution of viruses, pathogens, and other organisms of relevance in onsite and sewered sanitation systems.

The transport of solids and of the products of the reactions, such as inert compounds also need to be defined as a function of the rheology of faecal sludge and the process conversion processes such as solids degradation and the generation of inert and non-degradable products. Studies on soil mechanics and peat soils can be used for this purpose. Equally important is to study the transport and/or diffusion of gases (*e.g.* methane, carbon dioxide, hydrogen sulphide) into the different zones and layers (a solid-gas transport phenomena). This is mostly because the presence or accumulation of some of these gases (*e.g.* carbon dioxide and hydrogen sulphide) will affect the potential inactivation of pathogens in a direct or indirect manner (*e.g.* carbon dioxide by affecting the pH and hydrogen sulphide through a direct inhibition or toxic effect). Research already conducted on the transport and diffusion of gases in soils, peat soils and wetlands would support this future research (Armstrong *et al*., 2000; Aachib *et al*., 2002, 2004; Allaire *et al*., 2008). Understanding the transport and spatial distribution of pathogens and the generation and transport of key gases through the layers and zones of faecal sludge systems can contribute to studying potential strategies to enhance the inactivation of pathogens.

6.3.6 Onsite sanitation modelling: calibration and validation

For calibration and validation purposes, the same recommendations that apply to ASM can be followed. If the description of the performance shows that a major adjustment is needed (*e.g.* major adjustments of the kinetic parameters), the model structure and also the mass balances and data collection probably need to be revised. Sludge accumulation is the first aspect to be calibrated, followed by the most kinetically sensitive process (possibly hydrolysis or fermentation) and the rest of the kinetic processes. If the process performance and the quality of the generated flows have not been well predicted, uncertainty and sensitivity analysis can be conducted to assess which parameters have the strongest impact. Following an iterative step-wise process, the model is calibrated by adjusting the least possible number of kinetic parameters until it provides a satisfactory description of the performance of the system.

The model can be validated by assessing its capacity to predict the performance of the system using operational and environmental data from a different period than that used for the model calibration. It will need to be re-calibrated iteratively if it fails the validation step until a satisfactory validation is reached.

6.3.7 Onsite sanitation modelling: detailed characterisation

6.3.7.1 Faecal sludge characterisation and fractionation

As previously discussed, the use of dissolved oxygen meters, redox probes and microelectrodes (Sexstone *et al*., 1985) in vertical and horizontal directions, in combination with the conduction of aerobic and anaerobic experimental methods (Van Loosdrecht *et al*., 2016) using undisturbed core samples from sanitation systems and the characterisation at different heights of relevance, will be necessary in order to determine the extension and size of the aerobic and anaerobic zones. Once they are known, the faecal sludge needs be characterised and, more importantly for modelling purposes, it needs to be fractionated into the COD fractions of relevance. The fractionations required for aerobic models (Henze *et al*., 2000) and anaerobic models (Batstone *et al*., 2002) are different, yet to a certain extent similar from a biodegradability perspective (Ekama *et al*., 2007).

In view of the limited experience and information available concerning the fractionation of faecal sludge, further research needs to focus on the determination of the required fractions through the execution of experimental methods and, whenever possible, supported by elemental composition analysis following a structured and common protocol. Furthermore, it will be very important to carry out a characterisation and fractionation campaign in different countries and regions to reach a consensus regarding the most suitable and practical steps. This will be extremely useful to develop a suitable protocol for faecal sludge characterisation and fractionation similar to those developed in the past decades for activated sludge modelling (*e.g.* the BIOMATH, HSG, WERF, and STOWA calibration protocols) (Vanrolleghem *et al*., 2003; Langergraber *et al*., 2004; Melcer *et al*., 2003; Hulsbeek *et al*., 2002; Roeleveld *et al*., 2002).

To model the aerobic degradation of faecal sludge, a COD fractionation similar to that carried out by Lopez-Zavala *et al*. (2002, 2004a, 2004b) can be conducted using real faecal sludge. To determine the required aerobic kinetic parameters, a combination of respirometric tests (Ekama *et al*., 1986; Kappeler and Gujer, 1992; Spanjers and Vanrolleghem, 1995; Vanrolleghem *et al*., 1999) and activity tests can be executed (Van Loosdrecht *et al*., 2016). The information provided by these studies will contribute to obtaining a better estimation of the aerobic COD fractionation of faecal sludge and of the hydrolysis and degradation of faecal sludge under aerobic conditions.

Since most faecal sludge collection and treatment systems are anaerobic, the determination of the faecal sludge anaerobic fractions deserves special attention in order to apply ADM1. However, although ADM1 can be recommended as the most suitable model for faecal sludge modelling, there are two major interrelated challenges for its application in this field. First is the thorough fractionation of the feeding components required by ADM1, and second, as expected, is the rather limited research and information regarding the anaerobic fractionation of faecal sludge. Thorough ADM1 fractionation necessitates the determination of the (individual) compound concentrations (using specific analytical techniques) of soluble (S) components such as sugars, aminoacids, long-chain and fatty acids, as well as those of particulate (X) components such as composites, carbohydrates, proteins and lipids. Most of these parameters can be determined following the analytical methods described in Chapter 8. For modelling implementation, determination of large numbers of individual compounds is a serious disadvantage (Kleerebezem and Van Loosdrecht, 2006a, 2006b). It is a major structural bottleneck that has been observed in reviews of the implementation of the ADM1 model (Batstone *et al*., 2015). To overcome this bottleneck, certain approaches have been proposed: (*i*) to lump together the elemental composition of organic substrates using a limited number of widely available analyses (Kleerebezem and Van Loosdrecht, 2006b), (*ii*) to perform experimental methods to determine the anaerobic degradation kinetics needed to split the COD of a substrate into the input variables required by ADM1 (Girault *et al*., 2012; Poggio *et al*., 2016); and, when coupling aerobic models (*e.g.* ASM) with ADM1 for plant-wide modelling, (*iii*) to use interfaces to convert the aerobic fractionation of ASM models into the anaerobic fractionation of anaerobic models (Volcke *et al*., 2006; Nopens *et al*., 2009; Flores-Alsina *et al*., 2016). In previous efforts regarding faecal sludge modelling when ADM1 was applied, Wendland (2008) carried out a direct fractionation using specific analytical techniques. However, Elmitwalli *et al*. (2011) derived the required faecal sludge fractions from previous characterisation studies where the fractions were not directly determined (Elmitwalli *et al*., 2001; Kujawa-Roeleveld *et al*., 2003). To overcome these gaps, a suggestion is to carry out 'anaerobic' respirometric tests (Holliger *et al*., 2016) following a similar procedure such as that conducted by Girault *et al*. (2012) but using fresh faecal sludge. This approach will allow the faecal sludge anaerobic fractions and the hydrolysis kinetic rates required for the implementation of ADM1 to be determined. For this purpose, anaerobic respirometric tests need to be executed at different faecal sludge to anaerobic inoculum ratios. Ideally, anaerobic inoculum from real faecal sludge systems can be used but it could also be tested from different anaerobic sludge

digesters (in particular, from anaerobic digesters treating primary sludge which tends to resemble faecal sludge). In parallel, the determination of proteins, lipids and carbohydrates in faecal sludge based on standard analytical techniques (Rice *et al.,* 2017) and basic procedures (Kleerebezem and Van Loosdrecht, 2006b; Girault *et al*., 2012) can be used to support and validate the outcomes of the fractionation results. The results of the faecal sludge fractionation and its impact on faecal sludge systems modelling can be assessed by applying it to a real case or performing long-term SMA and BMP tests (Van Loosdrecht *et al*., 2016). The conduction of SMA and BMP tests (Holliger *et al*., 2016) will be useful to estimate the kinetic parameters of interest (hydrolysis, fermentation or acidification, acetogenesis and methanogenesis). However, the prediction of the anaerobic conversion processes will determine whether the conduction of continuous experiments is preferable, in particular to determine the faecal sludge hydrolysis kinetic rates (Batstone *et al*., 2009; Garcia-Gen *et al*., 2015).

According to Belia *et al*. (2009) and Nopens *et al*. (2014), there are four major locations of uncertainty that can severely affect the satisfactory calibration and validation of a model. They can be grouped as: (*i*) the inputs, (*ii*) the model, (*iii*) the model parameters and, (*iv*) technical or software aspects affecting the model. With regard to the inputs, it is important to characterise and fractionate the faecal sludge characteristics as accurately as possible and to provide a satisfactory description of the tanks and volumes. However, a major source of uncertainty is the variable generation of faecal sludge volumes, as pointed out by Brouckaert *et al*. (2013). The structure of the model and potential interfaces are another important source of uncertainty. The third group of uncertanties includes the feed model and hydraulics, and determining where the different aerobic or anaerobic zones exist, as they influence the need to use either an aerobic or an anaerobic model (ASM vs ADM1, respectively) and the interfaces required to couple the models. The last source of uncertainty is the one driven by software limitations (such as solver or numerical problems that interfere with a correct execution of the simulations). Overall, the first three sources of uncertainty can start to be analysed following the framework described in section 6.3.4.1, whereas the last one depends on the simulator or software used. In order to evaluate the uncertainty, different methods can be applied (*i*) to characterise and prioritise uncertainty by evaluating the quality of the data collected, expert elicitation, parameter estimation and sensitivity analysis, (*ii*) to increase the quality of the information by quality assurance, extended peer review and also involving the stakeholders and direct users, and (*iii*) to quantify and propagate uncertainty in the outcomes of a model (*e.g.* through the application of Gaussian error propagation, Monte Carlo simulation, among others). A detailed discussion of these methods and approaches goes beyond the scope of this chapter. Nevertheless, specialised publications on these topics can provide enough information and knowledge for their implementation to model onsite and sewered sanitation systems (*e.g.* Von Sperling *et al.,* 2020).

6.3.7.2 Inhibition and toxicity

Due to the stratification and predominance of certain processes over others (such as hydrolysis and fermentation over methanogenesis due to the differences in the growth of the microbial groups) (Van Lier *et al*., 2008; Pratt *et al*., 2012), the potential accumulation of ammonium and of (volatile) fatty acids with its associated drop in pH will probably lead to the inhibition of methanogenesis (Colon *et al*., 2015). ADM1 has inhibition functions to describe the potential inhibition caused by these compounds (Batstone *et al*., 2002). They will need to be assessed, validated and, if required, adjusted when treating and dealing with faecal sludge.

During the anaerobic degradation of organics, there is a potential risk that sulphate-reduction processes take place as a consequence of the human diet (Florin *et al*., 1993) or intrusion of water rich in sulphates (such as seawater in faecal sludge units located close to the coastline) (Van den Brand, 2015). Consequently, anaerobic sulphate conversion processes may lead to the generation of hydrogen sulphide (H_2S) which can inhibit methanogenesis both directly (since H_2S can be toxic to methanogens and other organisms) and indirectly (due to the

consumption of organics outcompeting methanogens) (Van Lier *et al*., 2008). Sulphate-reduction processes were not included in the original ADM1 (Batstone *et al*., 2002), but different extensions have since been developed and included (Kalyuzhnyi *et al*., 1998; Fedorovich *et al*., 2001; Barrera *et al*., 2013, 2015). A similar approach can be adopted to assess and describe the potential occurrence of sulphate-reduction processes in faecal sludge systems.

Moreover, the potential toxicity caused by cleaning and sanitising solutions used in toilets, external additives or other toxic compounds (such as motor oil, batteries or solvents) also needs to be taken into consideration. For this purpose, the protocol developed by Astals *et al*. (2015) to rapidly assess any potential inhibition or toxicity effect could be adapted and tested on faecal sludge.

6.3.7.3 Pathogen inactivation

The main objective of sanitation is the assurance of basic and safe public health. As such, the safe disposal of faecal sludge and the potential inactivation of pathogens is of major importance and deserves special attention. Different authors have studied and developed expressions to describe the inactivation of pathogens in different systems (see Table 6.6). However, to date, such expressions have been only marginally incorporated into mathematical models to describe the inactivation of pathogens in faecal sludge collection and treatment systems.

pH

pH has a major influence on the inactivation of pathogens. Extreme pH levels, either low (<4.0) or high (>9.0), result in satisfactory pathogen inactivation rates (Anderson *et al*., 2015). Mendonca *et al*. (1994) described how the pathogen inactivation observed at higher pH levels may be associated with the lysis of cells due to the disruption of the cytoplasmic membrane. Meanwhile, Russell (1992) proposes that, if the organisms cannot adjust their intracellular pH (which usually lies between a pH range of 6.0 to 8.0), at lower pH levels the accumulation of anions is responsible for the toxic effect of fermentation acids (*e.g.* acetic, propionic or butyric acids).

In treatment systems, pH is severely affected by the presence of acid-based systems and strong ions (Fairlamb *et al*., 2003). As such, the biological and physicochemical processes occurring in faecal sludge collection systems (or promoted by external factors such as co-digestion) (Riungu *et al*., 2018a, 2018b) or the addition of additives (Anderson *et al*., 2015; Riungu *et al*., 2018b) may lead to extreme pH levels that can enhance pathogen inactivation. For instance, the accumulation of acids (often interlinked to or influenced by a higher temperature) also led to a drop in pH during the (co-)treatment of faecal sludge in the studies carried out by Riungu *et al*. (2018a, 2018b). Overall, the decay rate of *E. Coli* reached up to 1.6 1/d with an accumulation of up to 16.3 g VFA/L at a pH of 4.9, whereas in a similar study (Riungu *et al*., 2018b), concentrations of non-dissociated VFA of up to 6500 mg/L led to a full inactivation of *E. Coli* and *Ascaris Lumbricoides*. Bina *et al*. (2004) investigated the removal of faecal coliforms, Salmonella and helminth eggs using lime treatment at pH 11 and pH 12. In the Philippines (Strande *et al*., 2014), disinfection was achieved after 30 min at pH 12, after 60 min at pH 11.5 and after 120 min at pH 11.

Magri *et al*. (2015) assessed the effects of pH in combination with concentrations of ammonia on the inactivation of adenovirus, reovirus and bacteriophages in faecal sludge. They observed that bacteriophages were more resistant than viruses. If the pH was higher than 8.9 and the concentrations of NH_3 reached 35 and 55 mM, the maximum time for a 3-log reduction was 35 days and 21 days at 23 °C and 28 °C, respectively. The expressions used to describe the inactivation processes were obtained by fitting the inactivation data to either an exponential decay or a lag-phase decay equation, respectively, as follows:

$$N = N_o \cdot 10^{-k \cdot t} \quad (6.1)$$

$$N = N_o \left[1 - (1 - 10^{-k \cdot t})\right]^{10n} \quad (6.2)$$

Where:

N is the final counting of bacteriophagues or viruses,

N_o is the initial counting of bacteriophagues or viruses,

k is the inactivation rate ($k = 1/t_{90}$),
t is the period of time,
t_{90} is the decimal reduction time, and
n is a parameter fitted in the regression that determines the lag phase.

It is important to underline that Magri *et al*. (2015) observed that if the biodegradable organics present in faecal sludge were hydrolysed and fermented to VFA, the pH decreased from 8.7 to 7.7. This affected the nitrogen speciation, reducing the concentration of NH_3 and consequently decreasing the inactivation effect of this compound. This indicates that if the inactivation effect of either high pH and ammonia or low pH and VFA is desirable, the hydrolysis and the fermentation processes need to be uncoupled otherwise they may counteract the inactivation effect between each other. As such, pH is a key factor that can be used and potentially enhanced (by exploring alternatives to adjusting the operating and environmental conditions through mathematical modelling) to maximise pathogen inactivation in faecal sludge systems. Interestingly, ADM1 (Batstone *et al*., 2002) has the required expressions to estimate the pH under anaerobic conditions and also has different expressions to take into account the inhibition of methanogens at different pH levels and with different VFA concentrations. Such expressions can be expanded to consider the inactivation effect of other parameters (such as ammonia) and also the addition of external additives (such as other acids, urea or lime) to provide a better pH estimation. After the addition of a state variable to describe the outcome of certain defined pathogens, together with their required pH inactivation rates, the estimation of the pH can then be used to assess the inactivation of pathogens.

Temperature

Temperature has been reported to be an important factor for pathogen inactivation (Watanabe *et al*., 1997). However, a thermophilic temperature range is needed (55-65 ºC) for an effective inactivation (Polprasert *et al*., 1983; Mills *et al*., 1992a; Watanabe *et al*., 1997). Koottatep *et al*. (2014) observed, in septic tanks operated at higher temperatures, a 3-log reduction in E. *Coli* at 50 ºC and even a 6-log reduction to a level of about 10 most probable number (MPN)/100 mL at 60 ºC. Their results were described with the modified Weibull expression:

$$\log\frac{N_t}{N_o} = -b_T \cdot t^n \qquad (6.3)$$

Where:
N_t is the number of microbial populations at any time,
N_o is the number of microbial populations at the initial time,
t is the contact time and b_T is a temperature coefficient, and
n is the Weibull coefficient.

In Equation 6.3, the b_T values of 1.36 and 1.71, and n of 0.26 and 0.41 were used to describe the inactivation rates at 50 and 60 ºC, respectively.

In another study, Lübken *et al*. (2007) described the inactivation of pathogens with a multiple regression expression in an onsite anaerobic system used for faecal sludge treatment. For intestinal enterococci removal, the following multiple regression term was proposed:

$$n_{IE} = 98.29 - 2.2\left(\frac{1}{HRT}\right)^2 + (0.031 \cdot T) \qquad (6.4)$$

Whereas to describe the inactivation of faecal coliforms the following expression was drawn:

$$n_{IE} = 98.29 - \left(\frac{1}{HRT}\right)^2 + (0.031 \cdot T) \qquad (6.5)$$

In equations 6.4 and 6.5:
HRT corresponds to the hydraulic retention time (in days) and T to temperature, ºC.

Similar to the study of Koottatep *et al*. (2014) who performed different studies in septic tanks at diverse temperatures, this study showed then considerable inactivation rates were only observed at a thermophilic temperature (55 ºC) and HRT longer than approximately 5 days. However, such a high temperature range cannot be easily generated in, or

provided to, most sanitation systems. It is generally those systems that enhance the composting process (such as Fossa Septica), that are directly exposed to sunlight (such as WSP) or engineered systems (such as digesters) are able to reach the required thermophilic temperature range that can lead to pathogen inactivation.

Fidjeland *et al*. (2015) modelled the inactivation of *Ascaris* eggs at different temperatures and high ammonia concentrations. For a given number of $\log_{10}$ reduction in viability (LRV), they estimate that the treatment time required to inactivate *Ascaris* eggs can be described with the following expression:

$$t = \frac{1.14 \cdot (3.2 + LRV)}{\left(10^{-3.7 + 0.062\,T}\right) \cdot NH_3{}^{0.7}_{Pitzer}} \qquad (6.6)$$

In the previous expression,

T is the temperature, and

$NH_{3,Pitzer}$ is the activity of the ammonia ion following the Pitzer method which makes use of the software PHREEQ.

A simplified method to estimate $NH_{3,Pitzer}$ is presented by Fidjeland *et al.* (2015) using a simplified Emerson-Pitzer conversion. This conversion makes Eq. 6.6 valid and applicable under some typical conditions found in real conditions (e.g. 8.3-9.5 pH, dry matter content up to 20%, NH_{TOT} between 5 and 2,000 mM, and for temperatures between 5 and 45 ºC). Similar to the description of pathogen inactivation by pH, certain expressions can be incorporated into ADM1 to describe the fate of certain defined pathogens at different temperatures.

Ammonia

Other studies have also focused on the inactivation possibilities of ammonia either present in the faecal sludge itself or after the addition of urea. Ammonia efficiently inactivates bacteria at pH levels between 9.0 and 9.5. It enters the cell membrane, increasing the internal ammonia concentration and causing the bacterial cell to pump out protons to maintain its optimum cellular pH, eventually resulting in cell death (Bujozek, 2001; Hill *et al*., 2013). Previous studies report a reduction in numbers of organisms, including non-spore forming bacteria, viruses and parasites through urea additions to manure and faecal sludge (Nordin *et al*., 2009; Magri *et al*., 2015). Fidjeland *et al*. (2013) hypothesises that the intrinsic ammonia present in urine has the potential to sanitise faecal sludge if the urine is concentrated and not lost by ventilation. They observed the inactivation of *Enterococcus faecalis*, *Salmonella typhimurium* and *Ascaris suum* eggs by ammonia between 5 and 28 ºC at ammonia concentrations ranging from 40 to 400 mM. *Salmonella* was fully inactivated after 2 days whereas *Enterococcus* reached a 5-log reduction between 13 and 110 days as the ammonia concentration increased from 19 to 243 mg NH_3/L. At 23-28 ºC, a 3-log reduction in *Ascaris* eggs was observed within 1 to 6 months depending on the ammonia concentration as described by the Eq. 6.6 (Fidjeland *et al*., 2015).

When ammonia is limited, the addition of urea and its subsequent hydrolysis to ammonia can lead to extreme pH levels and create a sanitising effect in combination with cell alkalisation by the ammonia released from the hydrolysis process (Fitzmorris *et al*., 2007; Anderson *et al*., 2015). Vinnerås *et al*. (2013) observed that, after the addition of 3% urea, *Salmonella spp*. and faecal coliforms were not detected after 5 days, *Enterococcus spp*. after 20 days, and viruses as well as viable *Ascaris* eggs were not detected after 50 days. ADM1 contains different expressions that describe the ammonia concentrations released from the hydrolysis processes of organics (Batstone *et al*., 2002). Moreover, by making use of the pH, the species of ammonium and ammonia can be calculated and with the help of inhibition expressions their effect on the anaerobic digestion process is taken into consideration due to their damaging effect on methanogenesis. Bearing this approach in mind, the ammonia concentrations can be estimated with the use of existing ADM1 expressions and they can be coupled to the inactivation expressions previously presented to describe the inactivation of different pathogens present in faecal sludge systems.

Lactic acid

Lactic acid bacteria (LAB) have the ability to convert carbohydrates to lactic acid (Gujer *et al*., 1986; Anderson *et al*., 2015). Lactic acid can penetrate the

cytoplasmic membrane in the associated form, resulting in a reduced intracellular pH and disruption of the trans-membrane proton motive force (Herrero *et al*., 1985). Also, lactic acid reduces the bulk pH of the surrounding medium, influencing the activity of exo-enzymes and membrane-bound enzymes. Ligocka *et al*. (2005) observed that *Salmonella spp*. and *E. coli* in sewage sludge were inhibited under both anaerobic and aerobic conditions with lactic acid. Soewondo *et al*. (2014), conducting laboratory experiments on faeces, observed a log reduction in total coliforms of log 4 to 7.5 after enhancing the lacto-fermentation process. Zhu *et al*. (2006) reported that in addition to reducing the pH in the bulk liquid, the key antimicrobial property of lactic acid is its ability to reduce the intracellular pH of bacteria. Anderson *et al*. (2015) satisfactorily inactivated *E. Coli* using lactic acid after the addition of sugars and inoculums of LAB in 7 days. Although LAB need to be inoculated in faecal sludge systems, the fermentation and production of lactic acid can be relatively easily introduced to ADM1 following a similar approach to the one used for other carboxylic acids (*e.g.* VFA) (Nielsen *et al*., 1991a, 1991b; Mercier *et al*., 1992; Spann *et al*., 2018) and for the description of pathogen inactivation in onsite sanitation systems.

Other pathogen inactivation equations

There is a vast amount of literature and research describing the inactivation of pathogens. However, the expressions that can be extrapolated and incorporated into mathematical models of onsite and sewered sanitation systems can be narrowed down to only those that contain, or are a function of, environmental and operating conditions that can be found or developed in these systems. As such, only those expressions that are a function of or dependent on the pH, temperature, dissolved oxygen concentration, and organic load are worth testing to describe the inactivation of pathogens. Practically all these equations are empirical and drawn based on laboratory, pilot or full-scale studies. Furthermore, some of the expressions are dependent and functions of different parameters depending upon the regression method or approach followed. Consequently, most of them have been developed following a 'black-box' approach without considering the actual (biochemical and physiological) inactivation mechanisms. Table 6.6 provides an overview of such pathogen inactivation expressions that could be incorporated into ADM1. For the description of the parameters in the equations, the reader is referred to the original source.

Table 6.6 Possible pathogen inactivation expressions developed for different wastewater treatment systems that could be incorporated into ADM1 to describe the inactivation of pathogens.

Pathogen removal expressions	Comment/remark	Reference
$e^{k_b} = 0.6351 \cdot (1.0281)^{T_w} \cdot (1.0016)^{C_a} \cdot (0.9994)^{BOD}$	Modified dispersion model expression applied to full-scale municipal WWTP in Brazil	Polprasert *et al*. (1983)
$K_b = K_{b,T} + K_{b,pH} + K_{b,BOD} + K_I$	Dispersion model equation applied to a pilot-scale municipal WWTP in Austria	Qin *et al*. (1991)
$K_b = 0.712 \cdot 1.166^{(T-20)}$	Completely mixed model equation applied to municipal plants in Kenya	Mills *et al*. (1992
$K_b = 0.5(1.02)^{Tw-20} \cdot 1.15^{(pH-6)^2} \cdot (0.9978)^{(BOD_5-100)}$	Plug-flow model expression applied to aerobic ponds in Jordan	Saqqar and Pescod (1992)
For coliforms: $K_b = 1.359 \cdot (1.087)^{(Tw-20)}$ For coliphages: $K_b = 0.439 \cdot (1.044)^{(Tw-20)}$	Plug-flow model equation applied to facultative ponds in Chile	Herrera and Castillo (2000)
$K_b = 0.019 \cdot (0.915)^{(Tw-20)} e^{0.171_m}$	Dispersion model applied to municipal WWTPs in France	Xu *et al*. (2002)
$K_b = K_{b,20} + K_{b,pH} \cdot pH + K_{b,DO} \cdot DO + K_{b,I} \cdot I) \cdot \theta^{(T-20)}$	Plug flow model equation applied to a laboratory-scale system in Belgium	Ouali *et al*. (2014)

6.3.8 Modelling applications, benefits and challenges

Depending upon the purpose, the application of models is meaningful during the entire lifecycle of the sanitation technology, including the design, construction, operation, and evaluation stages. Similar to wastewater treatment practice, there is a spectrum of possible use of models during the lifecycle of the onsite sanitation technology as shown in Figure 6.10. The wastewater treatment practice revealed that the most cost saving is possible when models are used in the early stage of the WWTP lifecycle, and similar expectations could be applicable to onsite sanitation technology as well. As discussed earlier in this chapter, the modelling goal determines the type and complexity of the model to be applied.

Figure 6.10 also depicts how the modelling complexity increases as the lifecycle of the sanitation technology progresses. The biggest savings are possible at the technology design phase because modelling helps to quantify scenarios at an early design stage. The quantification helps to speed up the decision-making process. High levels of uncertainty in the early design phases (*e.g.* due to faecal sludge composition) implies that large safety margins are needed (usual 150 to 200%), as such models can be simple (no calibration needed) and, during the design phase to invest in models and modelling work regularly pays back. Furthermore, practice shows that the highest financial risks are at the operational stage and modelling helps to reduce these operational risks (operational problems are often complex and more accurate models are required, *e.g.* ASM models in the case of WWTPs).

Reasons to introduce models in faecal sludge management at institutional level are: (*i*) to standardise the operation and management, control and quality assurance, (*ii*) to improve efficiency and reduce costs, (*iii*) to generate a knowledge base (organise process documentation), (*iv*) to improve internal and external communication (standardisation of information), and (*v*) to facilitate planning and decision making, etc.

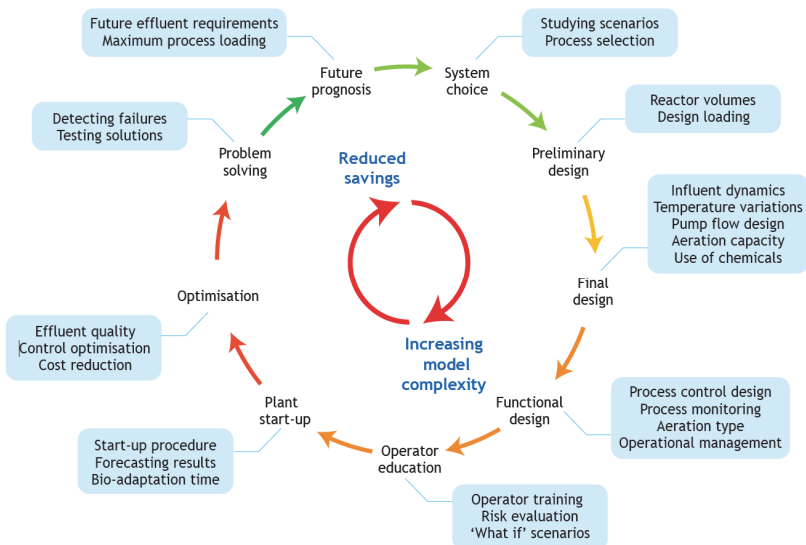


Figure 6.10 Modelling application at different stages of the sanitation technology lifecycle (adopted from Meijer and Brdjanovic, 2012).

Reasons for sanitation professionals to use models are: (*i*) to improve their work by better understanding the design and the process, (*ii*) to undertake regular training to update their skills and knowledge and introduce new and state-of-the-art technologies and approaches, (*iii*) to create a low-cost and safe platform for testing new ideas for improved operations and design, and (*iv*) to provide more efficient and improved decision-making and communication tools.

Success factors for using modelling in design can be summarised as: (*i*) following a protocol, providing realistic project planning and a practical approach, (*ii*) giving a 'bird's eye' view of the modelling project, (*iii*) defining clear modelling goals, (*iv*) keeping the model as simple as possible, and (*v*) using a standard calibration method.

However, one should be aware that by modelling, several bottlenecks may be identified such as: (*i*) choice of methods and software is important - a standardised approach is required, (*ii*) a different approach towards sanitation information systems is often needed, (*iii*) there is a continuous need to invest in education (life-long learning), and (*iv*) sharing of knowledge through a modelling platform, meetings, internet fora, and specialist groups. However, modelling practice from sewered sanitation shows that in general the use of models saves money, improves the quality of investments, is effective for management and decision making, and is an important asset for sanitation practice. Finally, the use of models in faecal sludge management is expected to have several main advantages such as: (*i*) cost reduction (especially at the design phase), (*ii*) improved management and quality control, (*iii*) optimal technological/process design using modern tools, (*iv*) the application of innovative approaches, and (*v*) the development of designs at low cost, rapidly and with confidence (Meijer and Brdjanovic, 2012).

6.4 OUTLOOK

Overall, at a micro-scale level (individual units), modelling of onsite sanitation systems can help to increase understanding about the conversions that take place in these units, contributing to improved design and operation of the onsite (and also indirectly, sewered) sanitation systems. This can be achieved by, firstly being able to describe the performance of the sanitation units to satisfactorily predict the quantity and quality of the faecal sludge generated and, secondly, based on these aspects, estimate adequate emptying and disposal practices as a function of the faecal sludge volumes and their characteristics. At a micro-scale level in onsite sanitation systems, modelling can also help to improve the design of the systems as well as their operation, enhancing, for example, the inactivation of pathogens (due to public health concerns) and increasing the generation of desired by-products (such as biogas or nutrient recovery as fertilisers).

With an increasing interest in the recovery of resources, mathematical modelling of faecal sludge might also be used as a tool to assess and develop innovative (biotechnological) practices and applications for the recovery of valuable or revalorised resources (*e.g.* methane, biodiesel, bioplastics, and nutrient-rich products) in a similar way to how it is being done in the wastewater treatment sector (Van Loosdrecht and Brdjanovic, 2014). This may be possible because the original 'raw material' (*i.e.* human excreta) is practically the same.

Moreover, by mapping and determining the type and number of sanitation systems that prevail in a region or area (in addition to the expected volumes and characteristics of the faecal sludge generated in each onsite sanitation system in accordance with the modelling studies), it is possible to estimate the overall and average faecal sludge characteristics and volumes generated in that specific region or area and to define and suggest the most appropriate practices and technologies for emptying, transporting, (co-)treating and disposing of faecal sludge. Better emptying practices and improved faecal sludge transportation to centralised plants can contribute to improving the handling of faecal sludge volumes and ultimately to achieving the goal of a CWIS approach. With a better knowledge regarding the number and types of faecal sludge systems available in a given location and considering their typical or average operating and environmental conditions, the most

appropriate faecal sludge treatment technologies or practices can be selected. For instance, faecal sludge with a high biodegradable organic content can be further treated under anaerobic conditions for biogas production whereas septic sludge with a low biodegradable organic content may only need to be dewatered or dehydrated prior to safe disposal. This also requires the development of mathematical models to describe the dewatering and dehydration of faecal sludge. Also, the faecal sludge modelling aspects and considerations described in this chapter can also be applied (see Table 6.5) to improve the required and selected faecal sludge (co-)treatment process.

This chapter primarily addresses approaches to modelling of onsite faecal and septic sludge containment and treatment technologies by making maximum use of the extensive knowledge gained during more than a century of research on wastewater/sewage treatment and more than three decades of experience of using biological wastewater and sludge treatment modelling. This analogy is possible and logical because of the fact that in both cases urine and faeces are the main raw materials that enter into the sanitation system, be it sewered or onsite, and that the combination of physical, chemical and biological processes is an essential component in the treatment in both cases. As the two sectors are presently rather polarised, such an extension enables further integration of sewered and onsite sanitation technologies at a system level, which is an essential step towards a city-wide inclusive sanitation approach. Therefore, this chapter focuses on the development of approaches on how to model the selection of the most common sanitation technologies for faecal sludge containment and onsite treatment, recognising the fact that this area of interest has the most complexity yet the least understanding of all the components of the urban sanitation chain.

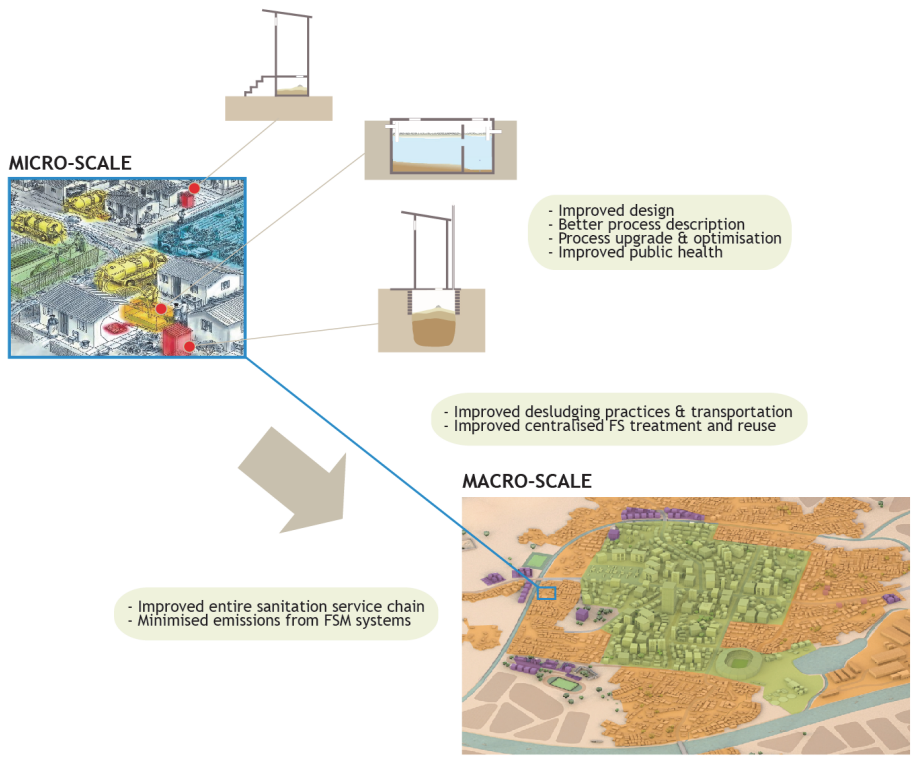


Figure 6.11 The micro- and macro-scale impact of modelling onsite sanitation systems (images adopted from Eawag).

Needless to say, this is just the tip of the iceberg concerning the modelling of FSM, whereas on the wastewater side the modelling of urban drainage and sewerage (Hvitved-Jacobsen *et al.*, 1998) and urban flood modelling (Price 2011) have advanced to the stage that can be combined with WWTP modelling and modelling of receiving waters (Hodzic *et al*., 2011, Brdjanovic *et al*., 2015) into an integrated urban sanitation model. Such an integrated model can be further extended towards a true CWIS model by the inclusion of the Q&Q model (Chapter 5) and combination faecal/septic sludge containment and treatment model as proposed in this chapter. Furthermore the onsite part of the model can be extended by collection and transport models (Anh *et al*., 2018) and models for onsite centralised treatment technologies (Strande *et al*., 2014). These models can be further integrated into a single holistic model at a city level where the challenge will be how to make all the necessary interfaces between different models so that models can properly communicate with each other. It is expected that such an integrated model will become available in the coming decade and that the first models will represent a steady state situation (*e.g.* seasonal or yearly average at the city level) and with further applications and developments, especially on the onsite sanitation side, a new generation of dynamic, real-time models will appear. However, even at this stage such an integrated dynamic model will not fully represent a CWIS model. For that it is necessary to include various business models (Strande *et al*, 2014) as well as the knowledge and application of behaviour models and citizen observatory approaches (Dreibelbis *et al*., 2013; Fritz *et al*., 2019) which can be seen as an attempt to extend CWIS modelling to Community and City Wide Inclusive Sanitation (CCWIS) modelling. These inclusions will increase the complexity of a CCWIS model; perhaps it will be necessary to create a simpler user interface that integrates more complex models working in the background with only some of the most essential features available to a regular (non-professional modeller) user. As the fundamental knowledge and number of models will continue to expand in the future it is to be expected that a new market for specialised modelling 'vendors' will be created and more complex modelling tasks will be outsourced to CCWIS modelling specialists.

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Figure 6.12 Data obtained from the field are essential for modelling of faecal sludge containment/treatment processes (photo; UKZN PRG).

Figure 6.13 The modelling skills are becoming increasingly important for the next generations of sanitation professionals. A modern approach of modelling and simulation of urban sanitation systems can be achieved through advanced level of competence and educational training, resulting in greater confidence, deeper insight and advanced knowledge. Modelling courses nowadays are offering to young sanitation professionals a chance to comprehend the scientific, technological and engineering principles of faecal sludge treatment and modelling (photo; IHE Delft).