sustainable sanitation alliance

literature review

working group on costs and economics of sustainable sanitation

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Introduction

Presented here is a preliminary summary of more than 90 publications on economic aspects of sanitation compiled by the staff of the gtz-ecosan programme. The aim of this work is to assess which topics are currently covered by the available literature and which information is further required. The reader is encouraged to contribute further input to be included in a second, enriched and expanded version of this review.

Most publications can be broadly classified as dealing with economics of sanitation (whether of small or large scale projects or on the global level) or with financial management of sanitation. The outline of this review therefore begins by characterisation of the various types of costs associated with sanitation, the common methodologies employed, followed by examples of costs of various systems and eventually a presentation of financial management schemes.

General note

All figures have been cited in the original currency as they appear in the respective source. Although the author tried to collect the most recent available data, one should keep in mind changes in the value of each currency. Special notice should be given to the frequently appearing US$ that has experienced considerable devaluation in latest years in comparison to many other currencies. Moreover, the reader should take into consideration the recent rise in market prices of energy and other relevant raw materials such as steel.

Estimating the costs and benefits of sanitation

Whenever a sanitation scheme is planned and technologies are chosen, costs always play a crucial role. It is important to distinguish between financial and economic costs. The first refers to costs borne by end users and financing bodies for construction and operation, while the latter refers to the overall costs and benefits borne by society (the nation, in practice).

Although calculating construction costs may be fairly straightforward, there is a considerable difficulty in generating exact figures on running costs for a typical planning horizon of 20-30 years (Sasse, 1998). One should also keep in mind, that variations in local conditions (topography, climate, socio-economic status, legislation etc.) can influence costs significantly (WHO, 2006). Other external impacts on health and environment are even more complex to assess and assign with monetary values. Nevertheless, although some assumptions must be made and perhaps several scenarios considered, it is always imperative to conduct an in-depth study to avoid non-functional systems and to minimise fiscal losses. Hence, detailed economic analysis is usually undertaken for large scale sanitation projects while smaller sanitation endeavours usually consider investment, operational and opportunity costs of capital (Franceys et al., 1992).

The following elements are normally calculated in financial *and* economic analyses:

1- Investment/capital costs: cover all materials, energy and labour expenses for the construction of all facilities and infrastructure necessary. Initial expenditures on equipment and land use are also included here. Relevant preparatory activities such as research and development, training, capacity building and promotion should be accounted for as well.

2- Running/recurring costs: refer to all materials, energy and labour expenses needed for the proper operation and maintenance (O&M) of the system. Reinvesting in replacing aged facilities represents a distinct form of running costs. Expenditures on monitoring and evaluation, quality control, issuing of permits and bureaucracy should be incorporated additionally.

3- Opportunity costs (of capital): reflect the potential lost profits if alternative investments would have been carried out with the capital used.

4- Revenues: applies when income may be generated by reducing greenhouse gases (through Clean Development (CD) or Joint Implementation mechanisms) or by using products such as biogas for energy production, sewage sludge as fertilizer in agriculture, treated water for irrigation etc.

The following elements are considered *only* in economic analyses:

1- Environmental costs/benefits: associated with pollution of soil, water and air, damages to biota and so forth.

2- Health costs/benefits: relates to expenditure on medical care.

3- Productivity: stems from the environmental and health impacts. Includes lost work and school days due to illnesses, losses to agriculture and industrial sectors from poor soil and water quality, losses to insurance companies and the tourism sector, and so on. Benefits from improved soil fertility or enhanced subsistence economies should be considered if observed.

4- Employment: generation of new work positions.

An important dimension that is often overlooked is the socio-cultural aspect. Enhanced sexual equality, convenience, security – all represent benefits to society. Economic (as opposed to financial) analysis considers the real costs for the economy – i.e. shadow pricing wages, subsidies, custom duties and taxes among others (Kalbermatten et al. 1982; Franceys et al., 1992; ADB, 1999).

Guiding principles of cost calculations

In general, calculations are made for the expected lifetime of the facilities, on an annual basis. For a fair comparison of different systems, it is advisable to choose a time reference that fits whole multiples of the various life spans (Franceys et al., 1992). When information is lacking, a rough estimation of O&M costs may be derived as a percentage of the investment costs (Toubkiss, 2000; Hutton & Haller, 2004; UNEP/GPA, 2004; Ilesanmi, 2006). Since maintenance costs of decentralised systems are a function of the number of users, it may be more appropriate to express expenditures per capita and/or per representative household size. This can be done be using the “average incremental cost” and the “total annual cost per household” techniques (Kalbermatten et al. 1982; Franceys et al., 1992; Wedgwood & Sansom, 2003).

Commonly employed methodologies

Net Present Value (NPV) – a means to assess the opportunity cost of money by adjusting future expenditures to their current monetary value. This is typically done by assuming constant, not inflated expenditures corrected by a discount factor. This approach tends to favour systems with lower investment costs even if ongoing costs may be higher (Franceys et al., 1992; Wedgwood & Sansom, 2003; Leal, 2004; Pinkham et al., 2004; von Sperling & Chernicharo, 2005; Mayunbelo, 2006).

Least-cost analysis – identifies the cheapest option amongst various sanitation schemes that yield similar results (Kalbermatten et al. 1982; Franceys et al., 1992). Some methodological variations enable ranking alternatives that deliver output of varying quality and quantity (ADB, 1999). The chosen least-cost alternative can be further investigated by a cost-benefit analysis.

Cost-effectiveness analysis – another technique, widely used for health care interventions, which expresses the ratio of costs and expected outcomes (not quantified in monetary terms). Larsen (2004) for instance, compared costs of various measures to prevent a child’s death from water borne diseases in developing countries. Brikké and Rojas (2003) point out that defining effectiveness and effects is not straightforward and may be subjective.

Cost-benefit analysis – perhaps the most widely used approach to evaluate sanitation systems. With this method, a ratio between the expected costs and benefits is calculated, enabling comparison of various sanitation schemes (for examples and discussions see Franceys et al., 1992; Hutton & Haller, 2004; Redhouse et al., 2004; Bajgain & Shakya, 2005; Prihandrijanti, 2006; WHO, 2006). Sasse (1998) on the other hand, argues that the cost-benefit method is not applicable for wastewater systems since sewage treatment does not produce profit. As stated before, it is difficult to assign monetary values to some benefits (Kalbermatten et al. 1982; Brikké & Rojas, 2003). Some methods that tackle these aspects are mentioned below.

Sensitivity analysis – a supplementary tool that scrutinises how changes in quantifiable variables (as opposed to the assumptions made) may influence a project's viability (ADB, 1999; Hutton & Haller, 2004; Bajgain & Shakya, 2005; Mayunbelo, 2006; Prihandrijanti, 2006).

Multi-criteria analysis – used to facilitate decision making on issues with multifaceted impacts that are specified and measured by quantitative and qualitative indicators (Ilesanmi, 2006; WHO, 2006; de Silva, 2007). It is therefore useful in considering indirect impacts on the environment, society and health (UNEP/WHO/UN-HABITAT/WSSCC, 2004). An extensive list of criterions and respective indicators for sanitation planning that considers these aspects have been presented by Kvarnstroem and Petersens (2004) and further elaborated in TUHH and TTZ (2006). Both sources add that the final selection of criteria should be done on a case to case basis, with regard to the unique conditions that prevail.

Willingness to pay (WTP) or contingent valuation (CV) – a method for estimating consumers' demand (maximal WTP) by surveys, and is therefore practical for financial planning for non-market goods such as water or sanitation services (Wedgwood & Sansom, 2003). Although some controversy exists on the reliability of WTP, it is recognised as a helpful decision making instrument when applied to use goods, as opposed to non-use goods such as biodiversity (Gunatilake et al., 2006).

Measuring health impacts can be facilitated by using combined indicators such as the DALY – Disability Adjusted Life Years (WHO, 2006). Redhouse et al. (2004) suggest a set of indicators with straightforward formulas to calculate impacts of water and sanitation projects such as improved health, avoided lost school days and gender issues although these methods may be oversimplified. In their report to the US Environmental Protection Agency, Pinkham et al. (2004) promote the use of an extensive "integrated wastewater planning" that includes, among other aspects, a review of methods to evaluate non-market goods and services. Hutton (2007) offers a broad and very systematic approach to appraise the economics of sanitation using a variety of costing techniques. He suggests that the relevance of each externality should be considered in each case and qualitatively evaluated if sufficient data is lacking.

Costs and benefits of different sanitation systems

Global overview

Proper water and sanitation services are strongly correlated with economical development (Euler et al., 2001; von Hauff & Lens 2001; SEI/UNDP 2006). Poor sanitary conditions result in great economic damages due to, among other things, spread of diseases, loss of school and working days, pollution of natural resources and reduced tourism (Briscoe, 1995; Wright, 1997; von Hauff & Lens 2001; Hansen & Bhatia, 2004; Hutton & Haller, 2004; Hutton et al., 2007a-c). The Millennium Development Goals (MDGs) are some of the most important measures, set with the aim to tackle these problems.

It is evident that providing decent sanitation services and achieving the MDGs in particular is very expensive. Some experts estimate an annual sum of US$ 11.5 billion for the implementation of the MDGs for sanitation, while others suggest a broad range of US$ 3.1-80 billion per year with respect to the technologies to be used (Evans et al., 2004; Toubkiss, 2006). Additional explanations for these disparities include different assumptions on population growth, baseline years and services level (Fonseca & Cardone, 2005). Even when considering the first and rather modest appraisal, current expenditure levels on sanitation (3 billion US$/year) cover merely a quarter of this sum (WHO/UNICEF, 2000). Two central challenges arise from these findings:

- Cost-efficient sanitation methods must be introduced in order to fill the gap (Kalbermatten et al., 1982; WHO/UNICEF 2000; Hutton & Haller, 2004; UNEP/GPA 2004; Fonseca et al., 2005; Rockstroem et al., 2005)

- Alternative financial schemes that make funds available ought to be explored (Briscoe, 1995; Steiner et al., 2003; Cardone & Fonseca 2003; UNEP/GPA 2004; Panesar et al., 2006; Trémolet et al., 2007b)

Different studies have demonstrated that fiscal gains from enhanced sanitation services surpass investments. Evans et al. (2004) for instance, found out that annual investments of US$ 20.5 million in Tanzania and US$ 96.7 million in Vietnam would yield benefits of US$ 15.4 million and US$ 66.7

million respectively for the health sector alone. Hutton and co-workers (2007c) calculated that the economical losses associated with lack of sanitation in Cambodia, Indonesia, Vietnam and the Philippines add up to US$ 9 billion yearly – about 2% of their combined gross domestic product (GDP). Of that sum, they estimated US$ 6.3 billion could be saved annually if proper sanitation and hygiene practices would be introduced. Additional work by Hutton and collaborators (Hutton & Haller, 2004; Hutton et al., 2007a) showed that benefit of various scenarios of global water and sanitation improvements (meeting at least the MDGs) exceed costs by a ratio range of 5 to 46. These gains are attributed mainly to time savings due to loss of work and school days. Complementary sensitivity analyses conducted by these authors have shown that even under pessimistic scenarios the benefit-cost ratios do not drop below 1. In a later version of this study (Hutton et al. 2007b) that considered only low-cost interventions for countries that off-track to meet the water and sanitation MDGs, the benefits of improved water supply were expected to be half of improved sanitation solutions, which included septic tanks, simple and ventilated improved pit latrines (VIPs). The higher average benefit-cost ratios of sanitation (9.1 for the MDGs and 11.2 for complete coverage) as opposed to water interventions (4.4 and 5.8 respectively) were ascribed to greater savings related to health and convenience time. The authors stressed however, that investment in sanitation is more expensive than water. For further information on economics of global sanitation, see the following publications: Briscoe, 1995; WHO/UNICEF 2000; Hutton & Haller 2004; Mehta & Knapp 2004; UNEP/GPA 2004; Rockstroem et al., 2005; Toubkiss, 2006; Hutton et al., 2007a,c.

Costs of conventional systems

It is difficult to deliver a concise account on the costs of conventional sewer systems as these are highly variable and site-specific. Therefore, such cost appraisals only offer a general idea on the magnitude and should be treated with caution. Fonseca (2007) estimated the annual per capita sewerage connection costs to lie between US$ 24-260, however treatment costs and expansion of infrastructure can considerably increase these costs. Fonseca adds that on-site solutions tend to be much more cost-effective, with annual per capita costs of US$ 11-54 for a simple pit latrine, US$ 10-172 for a VIP latrine and up to US$ 799 for a septic tank. Data provided by the WHO and UNICEF (2000) indicate that average per capita construction costs of sewer connections or septic tanks in developing countries are higher (US$ 115-160) compared with pour-flush toilets (US$ 50-90) and simple or VIP latrines (US$ 39-60). Annual per capita running costs of these systems were estimated by Hutton & Haller (2004), reaching in average US$ 5-13.4 for sewers, US$ 9-12.4 for septic tanks, and US$ 4-6.4 for latrines. Pinkham et al. (2004) carried out an in-depth cost analysis of on-site systems compared to centralised systems in the USA. They demonstrate through numerous case studies that investment, O&M and opportunity costs of decentralised systems can be more affordable, especially for small communities. Shilton and Walmsley (2005) present typical O&M costs of wastewater treatment methods that reduce organic matter and nutrient loads. Prices per 100 m^3 of sewage may be as low as US$ 2-10 (for mechanical treatment, flocculation and others) or as high as US$ 60-100 (for activated carbon absorption or ion exchange). Although, several methods are often incorporated to reach the desired effluent quality, Shilton and Walmsley argue that pond systems can be the single most cost-effective solution if land costs are not high. A comprehensive review of different treatment configurations was undertaken by von Sperling and Chernicharo (2005). Their findings confirm that although pond systems require much space, they can provide good performance for relatively low investment (15-40 US$/person) and recurrent costs (0.8-3.5 US$/person and year). Further information on conventional systems is given in the next section. Euler et al. (2001) compared the investment and recurring costs of pond systems, activated sludge and UASB reactors for a population equivalent of 50,000 with and without the expected returns from energy gains. Considering only BOD_5 as the discharge standard and assuming (among other things) land prices of US$ 25/m^2, the UASB technology (anaerobic treatment) was the most cost-efficient option. These authors argue that investment and running costs of anaerobic treatment are normally lower than those of activated sludge systems and also of pond systems if land prices are above US$ 10-12/m^2.

Randall (2003) depicts the economic damages of eutrophication of the Chesapeake Bay estuarine in the USA caused, among other sources, by sewage effluents. The enormous decline of some aquatic biota populations decreased the productivity of fisheries and resulted in the layoff of thousands of employees. Randall gives other examples for losses to fisheries due to nutrient pollution. A successful effort to improve the sustainability of 11 wastewater treatment plants in the Chesapeake Bay region (in the Virginia Peninsula) was the introduction of an enhanced biological phosphorus removal process. The nutrient rich sludge is composted and sold locally at a price of US$ 18.3 per m^3, and state regulations ensure its proper usage to minimise runoff pollution.

Alternative sewer systems (condominial sewers) with short length, small diameter pipes laid at shallow depths and gentle slope have been tested in Latin America. Savings of about 45% on construction and maintenance can be reached by condominial systems in Bolivia, while at Brazil even cutbacks of 60-80% have been reported (Watson, 1995; Foster, 2000). In an additional economic analysis, Foster shows that (assuming current tariff charges) a condominium sewerage connection in Bolivia can be 30-40% cheaper than a conventional one. Leal (2004) investigated construction and recurring costs of several sanitation concepts for the Al Moufty Al Kobra village in Egypt. She concluded that several source separation alternatives using small bore sewers would the cheapest options, as opposed to conventional and vacuum sewers.

Costs and benefits: conventional vs. reuse-oriented systems

Several authors have conducted comparisons between ecosan and conventional systems on a monetary basis. Panesar et al. (2006) acknowledge that few economic studies on ecosan are available and that most of them deal with pilot or demonstration projects, which are not reliable indicators for costs analysis. Such projects normally incur additional expenses on promotion, educational activities and production of unique system elements. The authors nevertheless claim that results indicate that ecosan systems have economic advantage over conventional ones, and stress the importance of using extensive multi-criteria analysis as a common basis for a fair comparison. Unfortunately, most of the authors that are cited in this section have only considered tangible costs and most benefits (with respect to environment, health etc.) have not been quantified.

UNEP and GPA (2004) developed the concept of the sanitation ladder, which ranks sanitation intervention levels according to their investment and recurring costs per person. Rockstroem et al. (2005) took this chart and added for comparison proposes, the costs of ecosan alternatives from projects worldwide (Table 1, below). Table 1 demonstrates that urban ecosan systems may cost the half of other centralised systems and even less as an on-site alternative. It should be kept in mind that these authors considered only the first year of operation in their calculations. Rockstroem et al. (2005) also estimate the per capita yearly costs of providing ecosan in developing countries to range between US$ 0.5 to 1 for rural areas, and between US$ 7 to 30 in urban settings. It however is not clear what these costs include and which kind of systems were considered. These sums are equivalent to less than 0.2% of domestic GDP in the developing world, rendering it affordable for households.

Shifting from the global perspective to specific case studies, Lechner and Langergraber (2004) investigated the cost effectiveness of three possible sanitation systems for rural villages in Austria. According to this study, a combination of low-flush and dry toilets with decentralised greywater treatment system are expected to have the lowest capital (€ 4,434) and operational (441 €/a) costs per household, compared with conventional sewerage and treatment plant (€ 8,790; 620 €/a) and with conventional sewerage system supplemented by urine separation, storage and reuse (€ 8,816; 700 €/a). These estimations take into account present subsidising practices without which households' investment and operation expenditures of both conventional systems would rise by about € 6,000 and 600 €/a respectively, making the alternative solution even more attractive. Oldenburg (2007) compared the costs of a conventional sewer system with a decentralised sequencing batch reactor to 5 hypothetical ecosan systems in a residential urban area of 4900 inhabitants in Berlin. He considered investment, reinvestment and running costs over a lifetime of 50 years with an annual interest rate of 3%. He found that one ecosan system would be cheaper than the conventional one – separate urine collection (emptied by lorries) and mixed collection and treatment of brown- and greywater in a sequencing batch reactor. Although all new sanitation systems would enjoy 13-17% lower O&M costs, their investment costs may rise by 25-60% due to 1.7-2.4 times longer pipework (particularly critical for 3-stream systems) leading to overall costs that are 3-13% higher. Additional sensitivity analyses conducted did not influence this trend. However, if served by another water utility operating outside of Berlin, which charges higher water and wastewater fees, all 5 ecosan concepts become considerably cheaper. In his doctoral dissertation, Schuetze (2005) analysed alternative water and wastewater systems (rainwater harvesting, greywater reuse, urine separation and vacuum toilets) for existing buildings in Hamburg and Seoul. He found that in Hamburg, user fees (as a function of investment and running costs) would be equal to the existing ones, while in Seoul they would be twice as high as current charges. He concludes that the structure of the fees and their subsidised price (in the case of Seoul) hinder the introduction of these systems. Hiessl and Toussaint (2004) have evaluated three urban (waste)water management models for two German cities. They report that investment and recurring costs of separation and reuse of wastewater streams may be only slightly more expensive than the traditional system. However, after applying a multi-criteria analysis the conventional solution was deemed least desirable.

Table 1: Sanitation cost ladder for conventional and ecological sanitation systems. Costs include initial capital cots and O&M for the first year of operation.

	Conventional Sanitation (sourced from UN Millennium Project, 2005; original source UNEP, 2004)		**Ecological Sanitation** (various sources see below)	
	Method	Estimated cost per person (USD) incl. operation and maintenance	Estimated actual initial capital cost per person (USD) and household incl. operation and maintenance (hh size is 4.5 unless otherwise given)	Method
Mainly urban	Tertiary wastewater treatment	**800**	340 (1190 per hh) (China, hh size 3.5)* (source: Dong Sheng EcoSanRes Programme)	Urine-diverting high standard porcelain dry toilet (indoor and multistory); piped urine system, dry faecal collection and composting, decentralised piped grey water treated using septic tank, and aeration treatment; local collection and transportation costs included
	Sewer connection and secondary wastewater treatment	**450**	330 (1500 per hh) (Sarawak)* (source: Mamit et al, 2005)	Conventional indoor toilet with sealed conservancy tank, black water collection by truck; local biogas digester; decentralised piped greywater treated using septic tank and vertical biofilm filter technique
	Connection to conventional sewer (assumed without treatment)	**300**	150 (675 per hh) (estimated)	Indoor dry single-vault urine-diverting pedestal toilet; decentralised piped greywater treatment using constructed wetland; local transportation included
Mainly peri-urban	Sewer connection with local labour (assumed without treatment)	**175**	88 (400 per hh) (South Africa) 25 (110 per hh) (Mexico, El Salvador, India, South Africa, Zimbabwe) (source: Morgan, 2005)	Dry single- or double-vault urine diverting squatting pan or pedestal toilet with permanent upper housing structure; greywater treatment using on site infiltration pit; transportation assumed as local labour
	Septic tank latrine	**160**	12 (55 per hh) (source: Lin Jiang, Nanning, Guangxi, China)	Dry single or double-vault urine diverting squatting pan or pedestal toilet (LASF or Skyloo) with permanent upper housing structure; greywater treatment and disposal onsite; local recycling
Mainly rural	Pour-flush latrine	**70**	8 (35 per hh) (West Africa) (source: Klutse & Ahlgren, 2005)	
	Ventilated improved pit latrine Simple pit latrine	**65** **45**	8 (40 per hh) (Zimbabwe, Mozambique) (source Morgan, 2005)	Soil composting pit with cement slab and simple upper housing structure (Arborloo or Fossa Alterna); grey water treatment and disposal onsite; local recycling
	Improved traditional Practice	**10**	3 (10 per hh) (estimated)	soil composting shallow open pit; soil added after each use

* Initial cost calculations are based on ongoing large scale pilot projects

Source: Rockstroem et al., 2005

The Ecosan Club (2003) compared expenses of flush toilets with a mechanical and vertical subsurface constructed wetland treatment with urine diversion dehydration toilets (UDDTs) and a horizontal subsurface constructed wetland system for a girls' school in rural Uganda. Investment, reinvestment, and O&M costs were calculated during a timeframe of 50 years with an annual interest rate of 8%. The results showed that the conventional system would be about 60% more expensive than the ecosanalternative, mainly due to the more compact wetland system and the additional expenditures on pumping required for the conventional system. The authors however estimated piping would cost the same for both variants which is doubtful and assumed for some reason 15 more UDDTs than flush toilets would be necessary.

According to Holden et al. (2004) urine diversion systems are more affordable compared to other contemporary solutions in South Africa. They have lower capital (ZAR 1,500) and running (0 ZAR/a) costs than VIP latrines (ZAR 2,000; 200 ZAR/a) or waterborne systems (ZAR 10,000; 1,200 ZAR/a). Zimmermann (2006) compared investment, reinvestment, and running costs of potential sanitation system in Syria over a period of 30 years. His results show that UDDTs or constructed wetlands could be € 5-20 cheaper per inhabitant and year than oxidation ditches (the currently favourable option). The UDDTs alternative remains undoubtedly cheaper than constructed wetland as long as the number of served inhabitants is below 50,000. More information on costs of wastewater treatment in Syria is reported by Mohamed (2006). The estimated construction (13.9 €/capita) and annual running (~1.0 €/capita) costs of a constructed wetland treatment serving 7,000 people are significantly lower than those assessed for the conventional alternative (200-250 €/capita and 50-100 €/capita, respectively).

Mayunbelo (2006) provides a comprehensive review on costs of VIPs (the common solution) and UDDTs for the entire population of Lusaka, Zambia's capital city. He found that both capital (€ 43.3 million) and running (2.8 million €/a) costs of UDDTs would be lower than VIP system (€ 47.7 million and 3.1 million €/a, respectively). The total costs for 10 years of operation in NPV would be € 59 million for UDDTs compared with € 65 million for VIPs. An additional sensitivity analysis revealed that the number of users per toilet would have a greater effect on investment than on running costs for both alternatives, while changes in the urine storage period would considerably influence investment costs of UDDTs (meaningless for VIPs). It is noteworthy that despite the rather short urine storage period (2 weeks) assumed for application in nearby agricultural lands, if at least some of the urine could be directly used by the households these costs would be reduced as storage is not necessary according to the WHO guidelines.

De Silva (2007) compared costs of UDDTs and VIPs in Accra (Ghana) based on multi-criteria analysis. Investment costs per capita are expected to be € 39 for UDDTs and € 27 for VIPs while annual operation costs (including transport, treatment and sale of by-products) were deemed similar (€ 2.2 and € 2.1, respectively). A sensitivity test demonstrated considerable influence of the potential selling price of urine and number of users per toilet. Nevertheless the overall benefits of UDDTs with regard to economical, social, environmental, health, institutional and technical aspects were higher. Abaire and Shane (2007) report the substantially lower construction costs of an Arbroloo (soil-composting latrine) in rural Ethiopia (US$ 5-12) compared with simple latrines (US$ 33-46) and VIPs (US$ 70-90), which led to a dramatic increase in the number of latrines constructed within the sanitation programme of the Catholic Relief Services Ethiopia. These authors emphasise that the ability to plant healthy and productive trees on the full pits motivates farmers to dig shallow pits in order to relocate the Arbroloo more often to maximise this benefit (see next section for fertilising value).

Ilesanmi (2006) examined six sanitation concepts for an area in a newly planned settlement (Kuje) in Nigeria, designated for 600 inhabitants. The results of his least-cost analysis indicate that the cheapest systems are low- or pour flush toilets with onsite Rottebehaelter treatment; low flush urine separation toilets and greywater separation with onsite storage, Rottebehaelter and constructed wetland treatment; UDDTs and greywater separation with onsite storage and constructed wetland treatment. The estimated overall investment and O&M costs (in NPV over 30 years) of these systems are in the range of € 2,000-2,300 and € 4,000-4,800 respectively, while the corresponding costs of conventional sewerage system with offsite treatment in ponds are expected to be approximately € 32,100 and € 214,000. It is however important to mention, that Ilesanmi did not calculate O&M explicitly but rather estimated them as 10% of the investment expenditures.

Olbrisch (2006) provides additional insights on costs of conventional and ecosan systems in four African cities. In Durban (South Africa) for instance, construction costs are in the range of € 331-398 for a VIP, € 398-597 for urine separation toilets (depending on the size) and about € 928 for septic tanks. According to Olbrisch, emptying of septic tanks and VIPs is done every 5-8 years and costs € 133, while annual costs of urine separation toilets are € 3.30-4.65, rendering them very attractive.

Investment costs per sewered connection are € 1,061-1,327 in the city centre (assuming one connection each 25m of sewer) and € 3,981-5,308 in the outskirts (connection every 500m). A fixed annual fee of € 53 per connection is levied on the users.

Etnier and Refsgaard (1998) have studied urine separation systems in rural regions and found them to be the most efficient sanitation method to avoid contamination from organic matter and nutrients and the most cost-effective one. Baten and Mels (2004) have compared the costs of advanced small waste water treatment plants with a source-separation system in rural regions of the Netherlands. While they estimate the investment costs of the ecosan system to be 15% higher, its running costs are expected to be 25% cheaper.

Costs and benefits of different ecosan systems

Some publications on planned or executed ecosan projects describe the costs of introducing the new systems. Details on costs of various ecosan systems from projects worldwide can be found on the project data sheets of the gtz-ecosan programme (GTZ 2005-2007). Unit prices of various models of urine separation toilets (GTZ 2006l), waterless urinals (GTZ 2005o) and compost toilets (GTZ 2006m) are also available in the technical data sheets. Additional data on average construction costs of urine separation toilets and Skyloo in Africa are provided by Jackson (2005).

The work of the Netherlands Development Organisation (SNV) has contributed much to the widespread use of domestic biogas systems at South-East Asia in recent years, which is now “invading” African markets. The report on phase 1 of the Vietnam programme (BPO, 2006) shows that with an institutional investment of US$ 2.4 million, 18,000 biogas plants that save households about 5 €/month on energy expenses and 1-1.5 of daily working hours (spent on biomass collection), can be built. Other economic benefits include improved indoor air quality in households, formation of a biogas market, reductions in CO_2-equivalent gas (some 35,000–55,000 tonnes/a), diminished use of wood and fossil fuels and enhanced gender equality. Additional information on the Vietnam programme is available in Heedge (2005) and in Teune and Ma (2007). As part of the Biogas for Better Life initiative 2 million biogas plants, half of which with attached toilets, are planned to be built in Africa by 2020. Winrock International carried out an in-depth cost-benefit analysis of the programme in Uganda, Rawanda, Ethiopia and for sub-Saharan Africa as a whole (Renwick et al. 2007). They estimated attractive financial and economical benefit-cost ratios of about 1.2–1.35 and 4.5–6.8 respectively.

A financial and economic analysis of the veteran biogas programme in Nepal has been published by Bajgain and Shakya (2005). They estimate the annual fuelwood savings (some 6,790 hectares of forest area) of the 111,400 units installed at US$ 4.8 million and 7.7 million litres of kerosene are saved annually at a value of US$ 2 million. They also found that household save 3 hours of work daily and US$ 21 annually on fertilisers through use of biogas slurry, and that 11,000 people have been employed during the project The SNV and gtz carried out feasibility studies to implement similar biogas projects in Bangladesh, Ethiopia, Rwanda, Burkina Faso and Tanzania (Nes et al., 2005; Eshete et al., 2006; Dekelver et al., 2005; Huba et al., 2007; Bos et al., 2007). Monetary benefits from reduced CO_2 emissions can be realised through Kyoto Protocol’s Clean Development Mechanism (CDM), which is further discussed in the next section. Snel and Smet (2006) offer additional information on construction and running costs as well as benefits from fuel generation and slurry use of domestic and community biogas plants in Bangladesh.

In India, public toilets with biogas plants have been constructed in different locations to serve poor and deprived populations. Hansen and Bhatia (2004) report that revenues from biogas facilities can recover the additional investment costs in 5-6 years, depending on whether gas will be used on- or off-site (for more detailed calculations see Bhatia, 2004). Financial assessments of investment and running costs of two other projects of public and school toilets are presented by Wafler and Heeb (2006) and Macwan and Heeb (2006). Both appraisals show high return rates (on investment recovery, see next section) due to the biogas and fertiliser usage. However, it is disputed whether profits derived from the type of crop grown (bananas, in the case of Wafler and Heeb) should be included in the calculation since it involves additional determinants and complicates the calculation. It is simpler to use a surrogate indicator, i.e. the market price of the replaced fertiliser, as suggested by Hutton (2007). China is the nation where domestic biogas has flourished the most. While 15 million Chinese households were using this technology by the end of 2004 (Nes, 2006), newer estimations suggest a figure of 20 million households (Heinz-Peter Mang, personal information). Nes presents the village of Shipai as a case study, where households save € 63.5 annually on energy, fertilisers, work and enlarged livestock. Other benefits mentioned are improved health, employment for technicians and more free time for women.

Investment costs of UDDTs, double pit (or urn) toilets, three-chamber septic tanks and biogas toilets in rural China were reported by Li et al. (2007). Exact figures are not available since financial support was provided as construction materials in some cases while in school projects aggregate costs that include urinals and wash basins were reported. Rough investment costs per household for biogas toilets are in the range of RMB 1,700-3,700, RMB 400-1,300 for UDDTs and around RMB 700 for double urn toilets. Other forms of costs as well as the various benefits were not included in this report. Construction costs of a standard UDDT in rural China were also reported by Kumar (GTZ, 2007g) to be RMB 750. However by using alternative construction materials or building the toilet indoors, expenditure on the superstructure have considerably reduced resulting in a unit price of RMB 300-500. Shayo (2003) reports the construction costs of a UDDT latrine in Dar Es Salaam, Tanzania to reach US$ 201. Like Kumar, Shayo mentions that these costs were lower when iron sheets, timber or thatch replaced cement in the superstructure, which constitutes up to 35% of the overall costs. According to UNDP/WSP-LAC/CENCA (2006), the capital investment of complete indoor lavatories equipped with a UDDT, a shower, a urinal, a sink and a treatment unit (combined settling tank with grease trap and a reed bed) for greywater and urine in Lima, Peru is US$ 603 and its typical monthly running costs are US$ 4.5 per household. Of this sum, the UDDT cost US$ 265, the urinal US$ 10, the treatment unit US$ 100 and the in-house piping US$ 30.

Prihandrijanti (2006) investigated three sanitation schemes for poor urban areas in Indonesia. The analysis showed higher cost-benefit ratios (NPV adjusted) of two different alternatives of source separation of wastewater streams with on-site treatment (urine storage, Rottebehaelter with vermin-composting and baffled septic tanks or anaerobic digesters) than that of a centralised (shallow) sewer system with off-site treatment in Imhof tanks. Additional sensitivity tests demonstrated that despite the influence of changes in variables, the ranking of alternatives remains the same. Slob (2005) conducted a detailed study on the logistical aspect of excreta collection and transport from UDDTs in a low-income community of 8000 in Delhi, India. Slob estimated that using a tractor trolley equipped with a pump would be the most cost-effective method to mobilise urine at an investment cost of about € 14,000-40,000 and total yearly costs (capital costs, O&M and labour) of € 29,000-97,000. The monthly costs per household would then be in a range of € 0.3-1 which may match the estimated WTP of around € 0.4. Slob expects that dried faeces would be most efficiently transported by an unmotorised tricycle, which investments costs were appraised at € 1,500 and total annual costs of € 5,300 that are equivalent to a monthly payment of € 0.07 per household.

Hutton et al. (2007c) estimated that Cambodia, Indonesia, Vietnam and the Philippines could generate US$ 271 million annually from biogas and fertiliser production if different ecosan systems would realise their potential market share. Rockstroem et al. (2005) computed the nutrient value of human excreta of the designated MDGs target group between the years 2003-2015, which could amount to an overall value of almost US$ 3 billion. This corresponds to a mean annual per capita value of US$ 0.5-5.5, which can contribute significantly to cost recovery in rural areas. In peri-urban and slum areas of Southern Asia this value can be as high as US$ 10 per person, making ecosan even more attractive. Etnier and Refsgaard (1998) estimated a similar fertiliser value of human excreta of about US$ 5 per person annually (Pinkham et al., 2004). Abaire and Shane (2007) evaluated the annual fertilising value of urine alone to be US$ 5 per capita which can replace 23 kg of mineral fertiliser (14 kg of diammonium phosphates and 9 kg of urea) for application on a 900 m^2 plot. Oldenburg (2007) compared the value of urine with chemical fertiliser taking into consideration the costs of agro-mechanical application in vicinity of Berlin. Although they estimated urine application to be almost 4 times more expansive, urine is still expected to generate a net benefit of 1.9 €/m^3. They also appraise the value of compost from faeces to be 60 €/m^3. Renwick et al. (2007) estimated the fertiliser value of biogas slurry to the economies of sub-Saharan Africa, assuming the low usage of mineral fertiliser in the region is negligible in financial terms, to be in the range of US$ 181-463 per plant and year.

The economic value of untreated urban wastewater reuse in the Guanajuato river basin, Mexico was investigated by Scott et al. (2000). They calculated annual savings of US$ 135 per hectare (US$ 18,900 for all 140 hectares) on commercial fertiliser and an annual value of US$ 252,000 for the water use. Costs of treating this sewage in a treatment plant are expected to be significantly higher than the current application. Health and environmental impacts were not surveyed in depth, and should be followed up. Initial analysis shows risk of eutrofication in the receiving water body, whereas accumulation of Nitrogen in groundwater and heavy metals in soils was deemed low. Although no clear indication of negative health impacts from pathogens were observed in the Guanajuato basin a detailed study was not undertaken.

A project in a suburb of Tufileh, Jordan, has set up greywater reuse systems for home gardens at 50 low income families (Faruqui & Al-Jayyousi 2002). An economic evaluation of the system found that average benefits equivalent to 10% (US$ 308) of the annual household income were generated through crop consumption and sale. The value of the recovered greywater as replacement for irrigation water corresponded to 27% of the average water bill. The mean net annual benefit (considering the associated costs) of four households was estimated at US$ 376 with a benefit-cost ratio of 5.3. It seems however, that the systems were in most cases comprised of basic diversion devices with no or very rudimentary treatment.

Gross et al. (2007) evaluated the economic feasibility of a greywater reuse for garden irrigation treated by a recycled vertical-flow constructed wetland for garden irrigation in southern Israel. With estimated monthly savings of US$ 20 on irrigation the investment cost (US$ 600) and annual O&M costs (US$ 100) are expected to be covered in 4 years time. Friedler and Hadari (2006) have studied the economic feasibility of urban greywater recycling for toilet flushing in Israel when using a membrane bioreactor (MBR) or a rotating biological contractor (RBC) technology. Reasonable investment costs for RBC systems (0.5% of an apartment’s price) were found for 5-storey buildings, while O&M costs can be covered by water savings in 7-storey buildings. MBR systems may only be economic under Israeli water prices when serving a cluster of buildings.

Funding

Although commonly viewed as a public service, many governments and municipalities typically only provide the initial capital for centralised urban sanitation systems that serve a fraction of the society, so that the majority of households must finance in-house and on-site installations on their own (Trémolet et al., 2007a). Some countries do not wish to relay on their national budget alone and allow the involvement of private companies in order to raise additional funds. Fonseca and Cardone (2005) present data on water and sanitation funding in developing countries from the year 2000. Local resources still comprised the lion’s share (65% from the public sector, 19% domestic private sector) followed by international aid (12%) and the international private sector (5%). While Trémolet et al. (2007b) provide similar figures, they underline that these are merely rough estimations. According to the WHO and UNICEF (2000), external support accounts for 35% of annual investment on sanitation and over 80% for water supply in developing countries (private investments were not accounted for). Both Trémolet et al. (2007b) and WHO and UNICEF (2000) mention that the degree of reliance on donor funds varies among the nations, the African countries tend to be the most dependant ones.

Planning of initial and future financial resources is required to ensure the construction, expansion and contentious operation. Pinkham et al. (2004) argue that principle issues of financial planning should be addressed before any cost evaluations. In order to guarantee the system’s sustainability, recovery of associated costs is to be attained. Should only financial or rather economical costs (and benefits) be considered? Some ecosan publications deal with the so called “financial internal rate of return” (FIRR), while others look at the “economical internal rate of return” (EIRR) as well. Examples for both applications can be found in: Pokharel & Gajurel 2003; Bajgain & Shakya, 2005; Dekelver et al., 2005; Nes et al., 2005; Eshete et al., 2006; Wafler & Heeb, 2006; Macwan & Heeb, 2006; Bos et al., 2007; Huba et al., 2007 and Renwick et al., 2007. Several authors (Franceys et al., 1992; Bockelmann & Samol, 2005; Cardone & Fonseca, 2003) underline the importance of identifying all costs and funding sources and provide a description of them. Nonetheless, even authors that look at the macroeconomic scale limit their calculations to few tangible cost factors. One exception is the work done by Renwick et al. (2007) that considered the value of domestic biogas in terms of heath, fertiliser use, reduced greenhouse gases and deforestation, among other things. They estimated a FIRR of 7.5–10% and an EIRR of 80–180% for the Biogas for Better Life programme in sub-Saharan Africa.

How should costs be covered and by whom? While it is generally accepted that end users should pay for running costs, the extent to which investment costs are to be paid by consumers, if at all, is disputed (Cardone & Fonseca, 2003). Sanitation fees or taxes levied on households are the usual ways to raise this money. Unsurprisingly, median sewerage tariffs are lower in developing countries, reaching 0.1-0.2 US$/$m^3$, while they may be 2 to 6 times higher in North America and Europe (WHO/UNICEF 2000). Hansen and Bhatia (2004) give some examples of tariffs in developing countries that are too low and therefore fail to ensure proper operation. The different types of tariffs, methods to determine tariff levels and means of collection are discussed by Brikké and Rojas (2003), Bockelmann and Samol (2005) and Cardone and Fonseca (2003).

Another common financial tool in sanitation is the utilisation of subsidies. Subsidies may encourage investments in sanitation by the private sector through custom duty exemptions or tax reliefs (Bockelmann & Samol, 2005). Subsidies may also be introduced to assist low-income populations that cannot afford to pay for sanitation services by themselves. There are ample examples of applying subsidies in (eco)sanitation projects (Wright, 1997; Hansen & Bhatia, 2004; Bajgain & Shakya, 2005; Dekelver et al., 2005; Nes et al., 2005; BPO, 2006; Eshete et al., 2006; Snel & Smet, 2006; Huba et al., 2007; Bos et al., 2007). However, improper design of subsidies may create market distortions and result in inefficient allocation of resources (Brikké & Rojas 2003). Against this background, there has been growing criticism on the misuse of subsidies in water and sanitation projects. Data from Briscoe (1995) demonstrate that the rich are those who benefit from subsidised sewerage in Latin America. Interestingly, he also observed that the gap increases in poorer countries, when subsidies are high and when sanitation is concerned. Wright (1997), von Hauff and Lens (2001) as well as Trémolet et al. (2007b) subscribe to this viewpoint too, mentioning that the better-off segments of society are often those benefiting from such subsidies. Participants in the evaluation surveys of the first phase of the Vietnamese biogas programme suggested that more emphasis should be given to impoverished households (BPO, 2006). In the second phase of the programme (perhaps as response to this criticism), three levels of subsidies that match households' economic status will be offered (Heedge, 2005). Another argument against subsidies is that they discourage social responsibility and lead to negligence since they decrease users' sense of ownership (Brikké & Rojas, 2003; Eshete et al., 2006). Mehta and Knapp (2004) mention for instance, the 1.7 million subsidised toilets built in rural Maharashtra, India, of which 43% are unused. This critique does not mean that subsidies need to be denounced, rather that their customary design be altered when necessary.

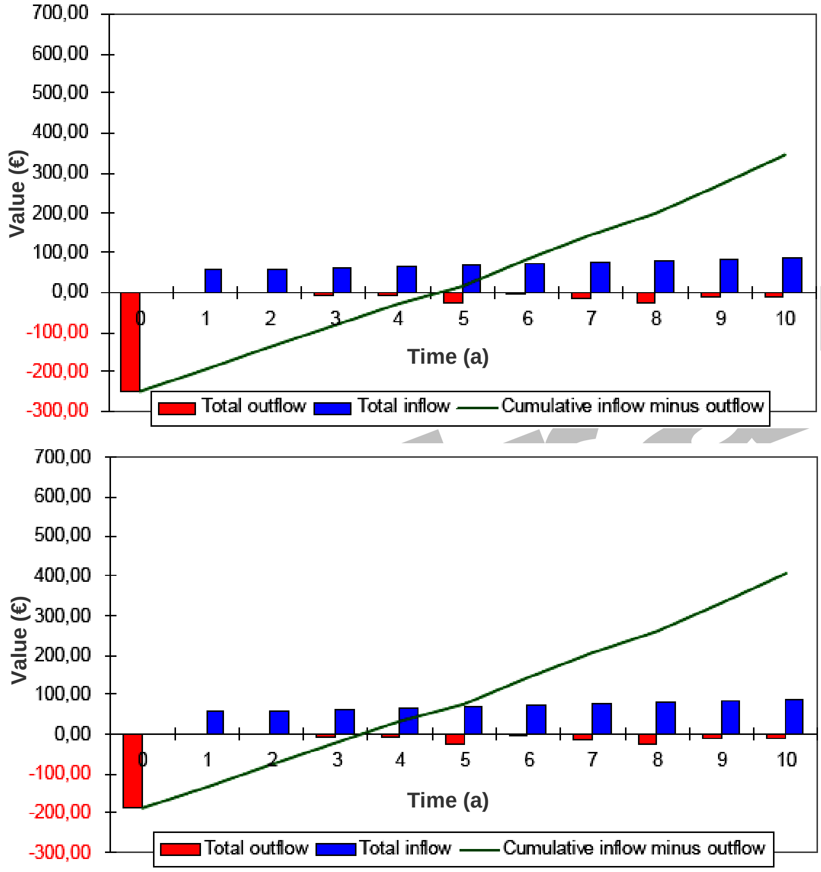
There is a growing awareness to the benefit of assessing and involving the direct beneficiaries in the financial planning of sanitation systems (Briscoe, 1995; Wright, 1997; Brikké & Rojas 2003; Hansen & Bhatia, 2004; Cardone & Fonseca, 2003; Mehta and Knapp, 2004). When consumers' opinion is not taken into account, it may result in low cooperation, unused toilets and costs that are not recovered (Wright, 1997). A demand-based approach in the planning process, in which stakeholders voice their needs and wants, is therefore promoted (Wright, 1997; Cardone & Fonseca, 2003; Brikké & Rojas 2003; Trémolet et al. 2007b). Kar and Pasteur (2005) recount the success of their Community Led Total Sanitation (CLTS) approach which was pioneered in Bangladesh and spread through South East Asia. CLTS drives communities to become aware of and responsible for their sanitary conditions, design and implement solutions by themselves without external subsidies of hardware. A financial facility may provide funds and financial management support for sanitation projects designed by local communities as in the case of the Community Led Infrastructure Financing Facility (CLIFF) in India (Trémolet et al. 2007b). Additional examples for such models are provided by Mehta and Knapp (2004), who urge to shift the financial weight towards promotion and resource leveraging activities. Some studies confirm the link between successful cost recovery and public participation expressed by a high WTP for specific sanitation technologies (Wright, 1997; Brikké & Rojas 2003; UNEP/WHO/UN-HABITAT/WSSCC, 2004, Trémolet et al. 2007b).

A financial instrument that has gained considerable popularity in recent years is microcredit. Micro-financing enables poor populations who do not qualify for normal credit prerequisites access to small loans, typically characterised by high interest rates and short reimbursement periods (Bockelmann & Samol, 2005). One acclaimed microcredit scheme innovated by Prof. Muhammad Yunus and his Grameen Bank (awarded 2006 Nobel Prize) in Bangladesh, enjoys a 95% payback rate through collective borrower responsibility, focus on female clientele and denial of collateral (Brikké & Rojas, 2003). Briscoe (1995) points out the increasing involvement of the Grameen Bank in rural water supply, lending some US$ 16 million for this purpose in 1993. Examples of micro financing in sanitation projects can be found in: Wright, 1997; Foster, 2000; Cardone & Fonseca, 2003; Kouassi-Komlan & Fonseca, 2004; Bajgain & Shakya, 2005; Nes, 2006 and Snel & Smet, 2006.

As mentioned before, an additional potential source of support of sustainable sanitation is available to developing countries through the CDM. Since CDM is relevant to projects that can demonstrate a reduction of greenhouse gases, the immediate candidates would be biogas sanitation projects. Here are some examples for projects that are CDM-approved (In Nepal - CDM Executive Board 2005a & 2005b), about to be approved (Vietnam - Heedge 2005; CDM Executive Board, 2005c) or consider CDM financing in the planning process (other SNV or gtz biogas projects - Eshete et al., 2006; Dekelver et al., 2005; Huba et al., 2007; Bos et al., 2007). Experience from different sanitation biogas programmes show that each biogas digester may save 2-5 tonnes of CO_2 annually (Yapp, 2006) and gains will depend on the carbon market prices and the type of credit gained (Bos et al., 2007; Huba et al., 2007). It is noteworthy that a certain amount of CO_2 reductions must be achieved in order to cover the expenses related to participation in the CDM (preparation, monitoring, certification etc.) which may exclude small scale projects (Yapp, 2006; Bos et al., 2007; Huba et al., 2007). Bos et al. (2007) and Huba et al. (2007) calculated yearly earnings per biogas plant in the first 7 years of operation of € 14-84 and € 6-60 respectively. Heedge (2005) estimated that the Vietnam project may generate revenues of € 46.3 million however since CDM revenues are only available upon delivery (after reduction has been achieved) and due to the uncertainty concerning the future commitments of the second Kyoto phase, only about € 16.7 million can be guaranteed at the moment. Heedge also calculated that through CDM earnings, reimbursement period can be reduced from 4 to 3 years and that after 5 years the FIRR and NPV improve from 2 to 13% and from minus € 20 to € 41, respectively. Figures 1a and 1b illustrate these findings.

Some attention has also been given to the issue of faecal sludge management (FSM). Steiner et al. (2003) investigated several models of financial structures to solve the problems of delayed and/or manual emptying of pits, indiscriminate disposal of sludge and nonexistent markets for biosolids. Their proposal is to alter money flows, so that desludging service-providers would be paid upon disposal to landfills through sanitation taxes and subsidies in addition to being paid directly by households. Snel and Smet (2006) present a detailed account on FSM at the city of Kumasi (in Ghana) and also suggest a shift in financial management and money flows.

Figure 1: The influence of CDM revenues on cost recovery of a fixed-dome biogas plant. The figures show the estimated financial in- and outflow with and without participation in the CDM within the Vietnam Biogas Project.



1a: Without CDM

- Simple payback in 4 years
- After 5 years:
  FIRR = 2% NPV = – € 20
- After 10 years:
  FIRR = 19% NPV = € 205

2a: With CDM

- Simple payback in 3 years
- After 5 years:
  FIRR = 13% NPV = € 41
- After 10 years:
  FIRR = 27% NPV = € 266

Source: adapted from Heedge (2005)

Summary and conclusions/the next steps

Monetary evaluation of sanitation systems involve several difficulties due to long-term planning horizon, impact of local conditions on costs, complexity of assessing externalities, variety in choice of methodologies and definition of system boundaries. Although sanitation services are deemed expensive, many studies demonstrate that investing in sanitation is economically worthwhile and can become more affordable when innovative technologies and financial mechanisms are introduced. The degree to which the aforementioned economic benefits could be realised (or losses mitigated), depends on the sanitation method and design. Nevertheless, although the main impacts on society are external, most evaluations of sanitation systems focus on the financial aspects alone. Therefore, one of the central issues that needs to be addressed is finding systematic scientific methods that assess the true value of sanitation interventions to society that can be widely acceptable. Another problem is the shortage of detailed economic data on innovative sanitation approaches and technologies.

The following modes of action are therefore recommended:

- Obtaining data from engaged organisations and companies, project managers and academics (i.e. through questioners/surveys, site visits, research etc.)

- Further developing and promoting the comprehensive evaluation criteria.

- Experimenting with the use of alternative financial schemes in sanitation projects.

- Pushing these ideas into the mainstream through cooperation with research institutes, political lobby, expansion of sustainable sanitation networks and so on.

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Acronyms and abbreviations

a	Annum
ADB	Asian Development Bank
BOD_5	5-day Biological Oxygen Demand
BPO	Biogas Project Office
CDM	Clean Development Mechanism
EIRR	Economical Internal Rate of Return
FIRR	Financial Internal Rate of Return
GDP	Gross Domestic Product
gtz	German Development Cooperation Agency
NPV	Net Present Value
MDGs	Millennium Development Goals
m^3	Cubic Metre
O&M	Operation and Maintenance
RMB	Yuan Renminbi (also abbreviated as CNY)
SEI	Stockholm Environment Institute
SNV	Netherlands Development Organisation
TTZ	Technology Transfer Centre Bremerhaven, Germany
TUHH	Technical University of hamburg-Harburg, German
UASB	Upflow Anaerobic Sludge Bed
UDDTs	Urine Diversion Dehydration Toilets
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UN-HABITAT	United Nations Human Settlements Programme
UNICEF	United Nations Children’s Fund
US$	United States Dollar
VIPs	Ventilated Improved Pit latrines
WHO	World Health Organisation
WSSCC	Water Supply and Sanitation Collaborative Council, Switzerland
WTP	Willingness To Pay
ZAR	South African Rand
€	Euro