Sanitation planning in developing countries

Added value of resource recovery

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Sjoerd Kerstens

Thesis

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No one needs morality when there isn't enough to eat

From: "The charge" on the album "Thunder and Consolation", New Model Army, 1989

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Chapter 1

The sanitation challenge

1.1 The sanitation challenge

The coming decades will bring profound changes to the size and distribution of the global population. The continuing urbanization and overall growth is projected to add 2.5 billion people to the urban population by 2050 to become 6.3 billion, with nearly 90 per cent of the increase concentrated in Asia and Africa (United Nations, 2014). These developments pose a challenge for food security and represent additional pressure on our food supply and on finite natural resources, such as phosphorus (Thornton, 2010; Gerbens-Leenes et al., 2010; Cordell et al., 2011).

Moreover, worldwide some 2.5 billion people do not have access to an improved sanitation facility and some 80 countries were not on track or made insufficient progress to achieve the Millennium Development Goals on sanitation (WHO & UNICEF, 2014). The anticipated population growth and urbanization will be an additional challenge in sanitation development in many countries, such as Indonesia that, despite modest improvement over the past years (WHO & UNICEF, 2015) are still in a poor state (ADB, 2013; Kearton et al., 2013).

The absence of well-functioning domestic wastewater and solid waste facilities is associated with a number of impacts:

First, discharge of untreated sewage can lead to adverse health effects in individuals exposed through contamination of drinking-water, contamination of irrigated crops or direct contact (Shuval, 2003). The World Bank's Water and Sanitation Program's (WSP) estimated that poor sanitation led to an economic loss of US$ 6 billion annually in Indonesia, equivalent to 2.3% of the national GDP (Napitupulu & Hutton, 2008). More than half of these costs were health related. Health conditions can be improved by implementing wastewater and solid waste interventions (Montgomery & Elimelech, 2007; Waddington & Snilstveit, 2009; Mara et al., 2010; Malekpour et al., 2013).

Secondly, discharge of untreated wastewater will increase the load of nutrients (nitrogen (N) and phosphorus (P)) and organic components, measured as Chemical Oxygen Demand (COD) and Biological Oxygen Demand (BOD), to the environment. This may result in eutrophication and low oxygen levels in (coastal) waters, impacting ecosystem functioning, and decrease revenues from fisheries and tourism (Hart et al., 2002; Fulazzaky, 2010; Suharyanto & Matsushita, 2011; Suwarno et al., 2013).

Thirdly, absence of wastewater and solid waste facilities may accrue socio-economic impacts, such as travel and waiting time for personal hygiene, loss of social capital and equity and decreased property values (Tayler et al., 2003; Alam, 2008; Fulazzaky, 2010; Winara et al., 2011; Hutton, 2013).

Finally, the value of valuable resources recoverable from wastewater and solid waste, such as energy, water, organics, nutrients, plastic and paper is being ignored in the absence of sanitation facilities or when applying conventional sanitation systems (e.g. landfilling of solid waste). Therefore, an aspect of sanitation development that receives increasing attention is the potential to recover resources from wastewater and solid waste (Lettinga, 2006; Almy, 2008; Aprilia et al., 2012; Thibodeau et al., 2014). The abundance of unmanaged solid waste and wastewater may result in an abundance of food and economic growth if these resources would be managed, recovered and reused in a sustainable way (Mcdonough & Braungart, 2000; Braungart et al., 2007; McDonough & Braungart, 2010; Kerstens et al., 2011). Besides the needs of households provided with sanitation systems ("front-end" users), also the needs of potential users of sanitation by products ("back-end" user) should be considered to foster long-term operational and financial sustainability (Murray & Ray, 2010a,b). Back-end users comprise among others agriculture (Schröder et al., 2011), aquaculture (Mungkung et al., 2013), horticulture (Aye & Widjaya, 2006), and plastic and paper processing industries (APKI, 2012; GBGIndonesia, 2013).

Therefore, the backlog in development of wastewater and solid waste facilities could be an opportunity. Adding the concept of resource recovery in the planning allows for direct introduction of a circular resource management, instead of developing a linear management system (Agudelo-Vera et al., 2011). Applying this concept on wastewater and solid waste management systems, involves a shift in paradigm (Guest et al., 2009; Larsen et al., 2009): emphasis is put on improvement of public health and environment as well as on valorization of waste, rather than on limitation of damage to public health and the environment applying a conventional approach, as is illustrated in Figure 1.1 (modified from Kerstens et al. (2011). In case (monetized) benefits of sanitation intervention exceed the implementation and operational costs, this may feed into advocacy efforts to raise funding from governments and households and may convince the private sector to invest in sanitation (Hutton, 2013).

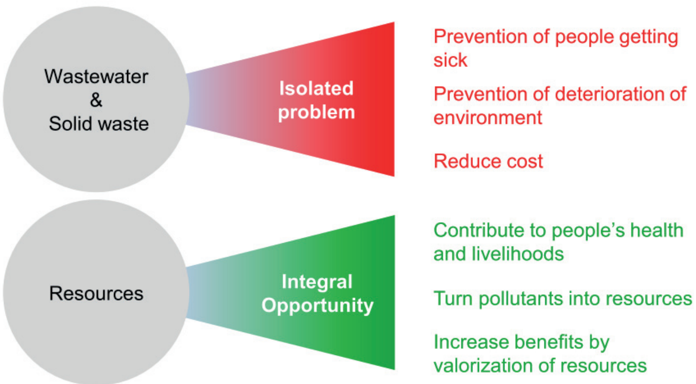

Figure 1.1 Change in perspective of wastewater and solid waste as an isolated problem to an integral opportunity by adding value to sanitation planning by resource recovery

1.2 State of the art of sanitation planning and resource recovery

The identified lack in sanitation development in developing countries has been attributed to inadequate sanitation regulatory frameworks and cross-sector policy coordination, rapid urbanization, low community awareness on the importance of sanitation, land availability, limited local capacity and knowledge to assure operation of facilities, and inadequate investments in sanitation systems (ADB, 2013; Kearton et al., 2013). In addition, the sanitation backlog has been attributed to the absence of a functional sanitation planning framework to support planning on spatial scales (village levels to national level) and temporal scales (short-term and long-term) (Tayler et al., 2003; Baum et al., 2013; WHO & UNICEF, 2014).

Planning of sustainable wastewater and solid waste interventions is a complex and cross-sectorial process. It requires integration of a variety of elements, such as (1) health, (2) technical, (3) environmental, (4) financial, (5) institutional, (6) socio-demographical aspects, (7) demand for sanitation by-products, and (8) socio-economical and welfare aspects (Table 1.1).

Because of the diversity of these elements, planning and evaluation cannot be achieved by one single method. A framework that combines several tools, methods and stakeholders to support decision or planning tasks is required (Thabrew et al., 2009; Mirakyan & De Guio, 2013).

The choice of an intervention, such as type of wastewater treatment system, may impact the health benefits, the receiving water quality, associated investment and operational cost, potential resource recovery and the required institution. Since these factors may mutually impact each other, a parallel evaluation provides significant benefits in a dynamic context (Pollack, 2009).

Table 1.1 Introduction of elements that require integration in sanitation planning

Elements	Example	Source
Health	The criteria for comparative evaluation of sanitation systems are the characteristics that are instrumental for fulfilling the objective of the system, such as health improvement.	Malekpour et al. (2013)
Technical	Appropriate water and wastewater technologies mean suitable and reliable technologies and depend on timing, the locality and socio-economic factors.	Ujang & Buckley (2002); Kujawa-Roeleveld & Zeeman (2006); Guest et al. (2009); Larsen et al. (2009)
Environmental	The impact of treatment system on pollutant load to the surface water bodies should be taken into consideration in system selection.	Tsuzuki (2006); Suwarno et al. (2013)
Financial	Planning of sanitation system requires insight in life cycle costs, Capital Expenditures (CAPEX) and operation and maintenance (Operational Expenditures (OPEX).	(Liang & van Dijk (2010); Ward (2012); Fonseca et al. (2010)
Institutional	Failure of sanitation systems often have an institutional nature, since policies, implementation and maintenance in the field of urban infrastructure lies on public institutions, that may not be capable or willing to come up with adequate policies and their enforcement.	Kvarnström & Mcconville, (2007); Van Buuren (2010)
Socio-demographical aspects	Relevant demographic and socio-economic projections, such as the rate of urbanization, and poverty should be considered when planning sanitation.	Loetscher & Keller (2002); UNEP (2004)
Demand for sanitation by-products	Insight in potential “back-end users,” and demand for the products of sanitation (e.g., treated wastewater or fertilizer) to motivate robust operation and maintenance of complete sanitation systems is needed.	Janssen et al. (2005); Murray & Buckley (2009); Murray & Ray (2010a); Linderholm et al. (2012); Diener et al.(2014)
Socio-economical and welfare	Poor sanitation causes significant losses to the national economy in terms of health, welfare and water quality.	Napitupulu & Hutton (2008); Winters et al. (2014); WSP (2014).

Thus, to evaluate the sustainability of a set of alternative sanitation systems, a framework for resolving trade-offs across spatial and temporal scales, and sustainability dimensions (social, environmental, and economic) is vital (Guest et al., 2009). Following abovementioned essential elements, this requires a framework that can combine and quantify:

1. Technical and financial criteria: Feasibility analysis of wastewater and solid waste systems under different residential (urban/rural) conditions;
2. Nationwide sanitation planning: Interpretation of this feasibility analysis in a (developing world’s) nationwide context considering planning targets, spatial and temporal population developments, financial, institutional and implementation implication;
3. Potential demand and supply of sanitation products: Spatial and temporal potential demand analysis for recoverable resources from wastewater and solid waste in relation to the potential supply;

4. Economic costs and benefits of alternative conventional and resource recovery-based wastewater and solid waste interventions.

The relation between these four quantifiable steps is shown in Figure 1.2. Thus, following planning targets set by a government as well as envisaged spatial and temporal demographical changes, the costs and benefits of sanitation systems are affected by the type of the selected system, implementation and planning path and supply and demand of recoverable resources.

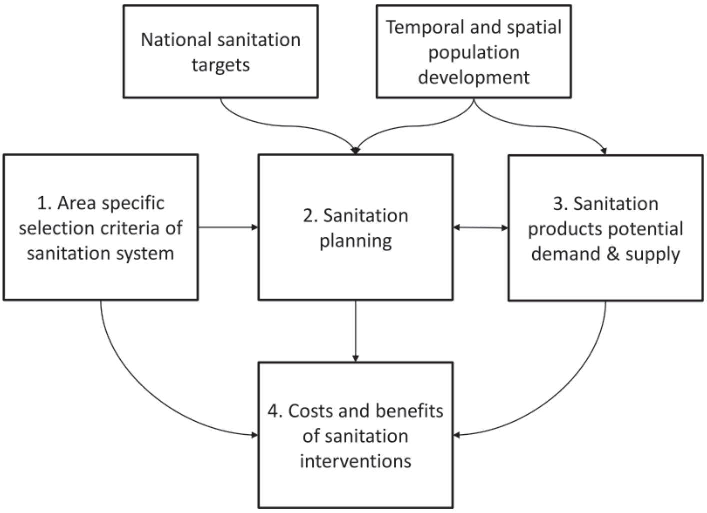

Figure 1.2 Framework to evaluate costs and benefits of sanitation systems in a nationwide planning context

1.2.1 Status on Sanitation systems and available selection criteria

1.2.1.1 State of the art of sanitation systems

A wide range of wastewater and solid waste systems, applicable for countries at different stages of their development, is available and has been described in international literature. An overview is presented below:

Wastewater treatment systems

Wastewater treatment systems may be categorized by the number of households served, distinguishing on-site systems (single household level), decentralized systems or community based systems (typically 50-200 households) and off-site systems (Ulrich et al., 2009; Kearton et al., 2013; Tilley et al., 2014).

The single pit is one of the most widely used on-site sanitation systems worldwide. Urine and water percolate into the soil through the bottom of the pit and wall, while microbial action degrades part of the organic fraction. To allow for continuous operation safer and easier emptying or enhance production of soil improver, the double ventilated improved (dry) pit latrine may be applied (Tilley et al., 2014). Another on-site system is the septic tank that removes 30-70% of suspended solids, COD and BOD and about 1 log of pathogens (Lettinga et al., 1991; Mgana, 2003). Often applied community based systems are anaerobic baffled reactors with an anaerobic filter or community bathing washing and toilets centers with a primary treatment (Wibisono et al., 2003; Ulrich et al., 2009; Reynaud et al., 2012a; Kearton et al., 2013). Off-site technologies comprise anaerobic systems, such as anaerobic filters and the Upflow Anaerobic Sludge Blanket (UASB) to remove COD and BOD and (Seghezzo et al., 1998; Said, 2000), pond based systems to remove COD, BOD and nutrients (EPA, 2002; Zhai et al., 2011; Mara, 2013), or activated sludge systems with (enhanced) biological nutrient removal (Brett et al., 1997; Baetens, 2001). Wastewater treatment technology developments are ongoing to further improve the reachable effluent quality and reduce the area footprint, such as membrane bioreactors (MBR) (Van Bentem et al., 2006), and, more recently, aerobic granular sludge (De Kreuk et al., 2005, 2007; Pronk et al., 2015).

A range of wastewater technologies have been described that may be used to produce or recover valuable resources from wastewater. Anaerobic technologies, like the UASB allow for production of energy from organic matter (Lettinga et al., 1993). Duckweed ponds have been applied to reduce the nutrient content from (anaerobically pre-treated) wastewater, while producing a valuable protein rich product (Van der Steen et al., 1998; Al-Nozaily et al., 2000; El-Shafai et al., 2007; Kerstens et al., 2009). Anaerobic technologies have also been successfully applied to digest the waste activated sludge, thereby reducing the amount of sludge that requires final disposal, while producing energy (Mata-Alvares et al., 2000). Valuable nutrients, like phosphorus and nitrogen, released during the sludge digestion can be successfully recovered using crystallization processes (Battistoni et al., 2002; Shu et al., 2006; Le Corre et al., 2009). Moreover, the use of microbial fuel cells may allow for simultaneous energy production and ammonia recovery from a nitrogen rich (urine) flow (Kuntke et al., 2012). Waste activated sludge as well as septic (fecal) sludge can be composted to produce a stable agricultural product (Haug, 1993; Veeken et al., 2003; Koné et al., 2007; Saveyn & Eder, 2014).

Above described technologies, except for septic tanks, typically treat a combined wastewater, comprising both (less polluted) grey water (from bathing, washing activities) and concentrated black (feces and urine) water. To enhance resource recovery, source separation has been proposed, since this allows for a higher recovery potential (Otterpohl, 2001; Lettinga, 2006; Larsen et al., 2009). Separate collection using minimal amount of transport water allows the production of a small volume of digested sludge and nutrients that can be used as fertilizer directly or after processing. Anaerobic treatment of black water, as a possible core technology for energy and nutrient recovery, may provide sufficient energy for the combined anaerobic treatment, nitrogen removal and phosphorus recovery (Kujawa-Roeleveld & Zeeman, 2006; Abu Ghunmi et al., 2011; De Graaff et al., 2011; Tervahauta et al., 2013). Compared to conventional treatment methods (e.g. conventional activated sludge systems), black water source-separation sanitation system may be significantly superior in terms of climate change, resources and human health indicators while having comparable cost, since they (may) produce energy and valorize resources (Thibodeau et al., 2014).

Solid waste systems

Municipal Solid waste management deals with the whole chain of (household) collection, transfer, transport and safe disposal or processing of solid waste (Meidiana & Gamse, 2010; Achillas et al., 2013). Solid waste comprises a mixture of materials, such as organic waste (e.g. vegetable, fruits, garden), paper, plastic, glass, textiles, hazardous components (e.g. batteries) (Saeed et al., 2009). In solid waste management, a distinction can be made between minimum interventions (all waste is landfilled) and interventions in which the amount of recoverable (organic waste, paper and plastic) waste that is disposed is reduced, reused and recycled (3R) (Antonopoulos et al., 2014). The application of 3R has been practiced at a decentralized scale as well as a centralized (e.g. city or region wide) scale (Pasang et al., 2007; Aprilia et al., 2012). The organic solid waste fraction can be treated using composting and/or anaerobic digestion (Haug, 1993; Veeken, 2005; Zhu et al., 2010) and may not only result in the production of a potential sellable compost and biogas, but also in a reduction on the amount of waste that requires landfilling (Norbu et al., 2005). Co-composting or co-digestion of waste activated sludge, fecal sludge and organic solid waste fraction can be considered as well and provides added value in terms of additional biogas production (Zupančič et al., 2008; Zitomer et al., 2008). In addition, household organic (kitchen) solid waste fraction can be directly co-digested with black water in a decentralized and source separated sanitation concept (Elmitwalli et al., 2006).

As a result of increasing demand for waste plastic, paper and metals, activities to recover these materials have been established fairly well in developing countries (including Indonesia), albeit typically by the informal sector (Saeed et al., 2009; GBGIndonesia, 2013; Sasaki & Araki, 2013; Chaerul et al., 2013). Different plastic solid waste treatment routes can be distinguished comprising primary (re-extrusion), secondary (mechanical), tertiary (chemical) and quaternary (energy recovery) schemes and technologies (Al-Salem et al., 2009). The recovery of materials such as waste paper, not only reduced the amount of waste to be disposed, but is also a relatively inexpensive input factor in the production of new paper products and may thus contribute to forest conservation (Berglund & Söderholm, 2003).

1.2.1.2 Identified knowledge gaps in system selection criteria

The selection of wastewater and solid waste systems and their related costs are an important element in planning sanitation systems in developing countries (Parkinson et al., 2014). Information on pros and cons of systems and technical (e.g. space, ground water table, reliability, energy consumption, resource recovery potential), financial (investment and operational costs) and social factors (employment, safety and public health, social acceptance) to consider in the selection of systems has been provided by Achillas et al. (2013) and Tilley et al. (2014). In addition, a wide range of comparisons and evaluations on wastewater and solid waste systems is available (USAID, 2006; WSP, 2011; Aprilia et al., 2012; Eales et al., 2013; Kearton et al., 2013). Finally, the causal link between population density and urban functions where a lot of people interact (e.g. shopping malls or Commercial Business Districts) and the increased occurrence of health issues and environmental problems due to absence of sanitation facilities has been well documented (Lasut et al., 2008; Mara et al., 2010; Wright et al., 2013; Gondhalekar et al., 2013). However, a combined feasibility analysis of wastewater and solid waste systems under different residential conditions and with different degrees of resource recovery is lacking in scientific literature (Ersoy et al., 2008).

There is a trade-off between the technical and financial performance of “high-cost”, better performing systems (in terms of pathogens, and organic and nutrients removal and low land requirements) on the one hand, and low-cost systems on the other (Rodriguez-Garcia et al., 2011; Malekpour et al., 2013; Mara, 2013). To link benefits (e.g. public health, the environment, resource conservation) of wastewater and solid waste systems to costs (implementation and operation and maintenance), the selection of sanitation system should consider both technical and financial criteria.

The technical feasibility analysis requires quantification of system performance, such as wastewater pollutant removal efficiencies (COD, BOD, N & P, micro-pollutants and pathogens) as well as quantification of consumption and production parameters such as, energy, sludge production, land use. This technical feasibility should include a sustainability analysis that identifies resource recovery potential from wastewater and solid waste such as energy, nutrients, water, organic fertilizer compost, duckweed, plastics, papers, metals (Tervahauta et al., 2013).

In a financial feasibility analysis, life cycle costs should be determined, since these include the capital expenditures (CAPEX) and operation expenditures (OPEX) of systems in the short and longer term. These take into account hardware (e.g. civil, electrical and mechanical works) and software (e.g. studies and design) costs, operation and maintenance, capital maintenance, and the need for direct and indirect support, including training, planning and institutional support (Fonseca et al., 2010; Starkl et al., 2012).

1.2.2 Sanitation planning

1.2.2.1 Current types of sanitation planning frameworks

Several options for sanitation planning support exist, such as frameworks, models, toolkits and software programs. These may differ in target group, scales (community to national level), degree of participation and complexity (Törnqvist et al., 2008). The involvement and participation of end-users as decision makers using basic tools (e.g. checklists) is an often applied approach for developing countries and allows for location specific selection and management of sanitation systems (Ulrich et al., 2009; Van Buuren, 2010; Parkinson et al., 2014). Other planning support have a high degree of complexity, depend on complex software tools, have a low focus on participation, and are resource demanding with regard to time, money and competence (Törnqvist et al., 2008). The use of Multi Criteria Decision Tools (MCDS) is a common tool for solid waste management strategies. MCDS are mostly dominated by cost and environmental impacts of alternative strategies, whereas technical (reliability, feasibility, applicability) and social (employment, safety and public health, social acceptance) criteria are far less considered (Achillas et al., 2013).

Planning tools considering the reuse or recovery from sanitation products are scarce (Murray & Ray, 2010b). An example is the Design for Service (DFS), comprising a five-step planning approach that results in a site-specific, reuse-oriented sanitation scheme. DFS is locally tailored to specific users and specific economies and therefore requires local expertise and a significant role for user participation and input (Murray & Buckley, 2009; Murray & Ray, 2010a).

The Service Delivery Assessment (SDA) of the Water and Sanitation Program (WSP, 2014) provides a nationwide budget plan and consists of (1) a review of past water and sanitation access, (2) a costing model to assess the adequacy of future investments, and (3) a scorecard that allows diagnosis of bottlenecks along the service delivery pathways.

1.2.2.2 Identified knowledge gaps in sanitation planning frameworks

Despite these available sanitation planning support tools, no framework could be identified that allow a planner or policy maker to quantify costs and benefits and resolve trade-offs across spatial (village to national level) and temporal (short and long-term) scales, and sustainability dimensions (social, environmental and economic):

Firstly, existing frameworks typically focus on specific population groups, distinguishing urban, rural or poor, non-poor communities (Törnqvist et al., 2008; Mehta & Movik, 2010; Sijbesma, 2011). However, within a country these population groups (co)exist. Consequently a framework should address all groups, while considering that these population groups likely have different (i) access to sanitation facilities (WHO & UNICEF, 2014; WSP, 2014), (ii) future access targets (e.g. urban and rural) (Bappenas, 2014), (iii) implementing agencies (rural implementation typically through the Ministry of health, such as in Indonesia or Lao (Mehta & Movik, 2010; ODI (Overseas Development Institute), 2011); urban implementation through the ministry of public works or construction, such as in China and Indonesia (Yan et al., 2006; WSP, 2014) and (iv) support needs (e.g. financial needs of poor communities) (Sijbesma, 2011).

Secondly, existing sanitation frameworks typically focus on one type of sub-sector (e.g. wastewater or solid waste intervention) or one type of solution (e.g. decentralized wastewater systems) only (Aye & Widjaya, 2006; Pasang et al., 2007; Kraemer, 2010; Henriques & Louis, 2011). Because both sub-sectors (wastewater and solid waste) aim to improve public health and the environment, they should be addressed and solved simultaneously to achieve the desired quality of life (Ersoy et al., 2008; ADB, 2013).

Thirdly, available sanitation planning frameworks may be too complex to apply on a nationwide scale or require too many detailed location specific information (e.g. groundwater and soil conditions, accessibility, proneness to flooding) such as the Quantitative Microbial Risk Assessment (Surinkul & Koottatep, 2009), SANEX decision support system (Loetscher & Keller, 2002) or the reuse oriented DFS approach (Murray & Buckley, 2009). On the contrary, other planning tools, such as WSP's SDA can provide a sufficient general insight in sanitation requirement, but are too general and lack a clear wastewater system selection and corresponding basis for costing as well as lack elaboration of the solid waste sector.

Fourthly, urban infrastructure investments are typically strategic and long-term in nature and have important spatial implications. Despite the availability of Geographic Information Systems (GIS) that may support regional priority setting and create awareness on the required implementation in land use planning activities (Quaye-Ballard & An, 2010; Coutinho-Rodrigues et al., 2011; Gondhalekar et al., 2013), this feature is not frequently linked to the output of a sanitation planning tool.

Finally, a planner should be able to define implementation and operational budgets per responsible institution (Mara et al., 2010; Winters et al., 2014). To scale up sustainable sanitation support it is important that financial support is directly channeled to implementing institutions and combined with capacity building and technical assistance (Iyer et al., 2005). Louis & Magpili (2007) developed a model that provides a disciplined but simple process for reducing persistent deficiencies in sanitation service capacity, but is designed for a community level and lacks the possibilities to scale up to a nationwide allocation.

Thus, a comprehensive framework that (1) directly links a government policy to a nationwide long-term planning and budgeting and corresponding allocation to responsible implementation institutions, (2) includes all existing population groups, (3) describes both sanitation (wastewater and solid waste) sectors and, (4) can visualize the implementation in GIS, is missing.

1.2.3 Analysis of potential demand and supply of sanitation products

To allow for valorization of resources recovered from wastewater and solid waste, it is essential to understand the demand for resources that may be recovered from wastewater and solid waste.

A distinction can be made between the potential demand and the market demand (Lyneis, 2000). Demand for resources depends on a number of factors. First of all, it is affected by population developments. For example, a growing population requires more food production, which may accrue an increased demand for (recoverable) fertilizers (Cordell et al., 2011). However, the market demand also depends on the availability of alternative resources and the recovery costs compared to prices of competitive resources (Cordell et al., 2011; Saveyn & Eder, 2014). The market demand is further affected by the quality and safety of produced products (Snyman & Vorster, 2011; Raschid-Sally, 2013), which may depend on the source and level of hygienization of sanitation by products (Koné et al., 2007) or raw materials (e.g. plastics) (Lazarevic et al., 2010). The market demand for resources has further been associated with the level of economic development, with "richer" countries showing a higher recovery rate than middle or low income countries in the case of paper recycling (Berglund & Söderholm, 2003). The market demand can further be affected by planned policies or activities (e.g. subsidies, campaigns) (WHO, 2006; Cordell et al., 2011).

The potential demand for resources provides a better indication than the (current) market demand for the possibility to convert to a circular economy (Agudelo-Vera et al., 2011), since it fully recognizes the inherent business value of recoverable resources (Braungart et al., 2007).

A method to visualize the potential demand and supply for resources is a Material Flow Analysis (MFA). An MFA maps fluxes of resources, such as organic matter, nutrients, water and recyclables (plastics, paper, metals) that are used and transformed as they flow through processes within specific system borders (Brunner & Rechberger, 2004). The MFA has been applied to analyze the potential for resource recovery from wastewater and solid waste and linking it to agricultural production and food security. MFA has been applied in cases in Ethiopia analyzing N&P and organic matter flows on a village level (Meinzinger et al., 2009), Thailand for water reuse schemes on a city level (Surinkul & Koottatep, 2009), China for phosphorus flows on a city level (Qiao et al., 2011), Indonesia for N, P and waste stream on a community level (Ushijima et al., 2012) and compost production and demand flows on a European level (Saveyn & Eder, 2014) .

The potential supply of resources follows from number of people served by wastewater or solid waste system that allows for a recovery of part of the incoming flow. Several studies have determined the amount of resources that can be recovered, such as phosphorus and energy (Zeeman & Kujawa-Roeleveld, 2011; De Graaff et al., 2011; Mihelcic et al., 2011), proteins (Van der Steen et al., 1998; Cheng & Stomp, 2009) or plastics or paper or small recyclable fractions like glass, textile or metal from waste streams (Berglund & Söderholm, 2003; Aprilia et al., 2011; Chaerul et al., 2013).

Quantification of the demand for resources that may be recovered from wastewater or solid waste requires and analysis of agricultural and consumer goods markets. The former group may comprise fertilizers (e.g. phosphorus) (Cordell et al., 2011), soil improvers such as compost (Veeken et al., 2005), proteins, such as duckweed or algae as a feedstock for aquaculture (Islam et al., 2004), biofuels from algae or duckweed (Cheng & Stomp, 2009; Adenle et al., 2013) and water (Janssen et al., 2005; Murray & Ray, 2010a). Solid waste may include organics, paper (Berglund & Söderholm, 2003), plastics (Al-Salem et al., 2009), metals and other valuable recoverable fractions such as glass (Kang & Schoenung, 2005). The resource demand analysis requires evaluation and insight in their potential uses:

- QUEFTS (quantitative evaluation of the fertility of tropical soils) may be used to evaluate the potential to use nutrients or organic matter from wastewater for soil fertilization for crop production (Janssen et al., 1990). QUEFTS allows for a detailed evaluation on nutrient requirements correcting crop demand for losses, nutrient accumulation on soils and ratio between nutrients (N, P and K) (Janssen et al., 2005). For long-term analysis, assuming an equilibrium between crop specific uptakes and losses would justify the possibility to use static fertilization rates independent of soil conditions as applied by Syers et al. (2008). Thus, based on specific nutrient and organic soil demand of crops the potential nutrient demand may be determined per type of crop;
- Wastewater treatment applying duckweed and/or algae ponds have been widely practiced as an efficient (nutrient removing) treatment step (Van der Steen et al., 1998; El-Shafai et al., 2007). Both duckweed and algae can be harvested and applied as protein rich resource for animal feed stock, organic fertilizer or biofuel production (Journey et al., 1993; Islam et al., 2004; WHO, 2006; Cheng & Stomp, 2009; FAO, 2010; van der Spiegel et al., 2013). Duckweed, as a protein rich source, is also expected to enter the European feed and food market as replacers for animal-derived proteins, but its use is subject to European Law (van der Spiegel et al., 2013). Studies for biofuel production from algae and duckweed show their promising potential, but also reveal that they are still in an early stage of development and not yet ready for full scale implementation in a development world's context (Adenle et al., 2013; Verma & Suthar, 2015);
- Potential demand analysis methods for end-of-life-product consumer goods, such as plastics, glass, paper and metals are still subject to much debate in literature. Their demand depends, among others, on development of recovery technologies, economic developments, sustainability policies and (international) trade patterns (Van Beukering, 2001; Berglund & Söderholm, 2003; Kang & Schoenung, 2005; Al-Salem et al., 2009; Lazarevic et al., 2010).

Demand forecast are subject to uncertainties and depend on the type of the market. Whereas for agriculture and food demand trends forecast can be made based on population forecast and GDP (Gross Domestic Products) (Tilman et al., 2011), more dynamic industries (e.g. consumer goods) rely also on external factors, such as macro- economic developments including fuel prices, manufacturing capacity and regulations (Lyneis, 2000).

Despite the availability of described methods and findings, a comprehensive framework that includes recoverable resources from both wastewater and solid waste and allows for a nationwide temporal and spatial demand forecast is lacking.

1.2.4 Cost and benefits of sanitation interventions

1.2.4.1 Existing methods to determine benefits of sanitation interventions

Policy makers should understand the combined potential outcomes (benefits) of major implementation (costs) before making choices (Ward, 2012). The Benefit Cost Ratio (BCR) describes benefits of intervention (e.g. health, social, resource recovery) relative to its costs (implementation and operation) and can be used to evaluate sanitation interventions (Alam & Marinova, 2003; Almy, 2008; WHO, 2012; Hutton, 2013). To identify the total costs and benefits of this implementation, monetization of both use values (e.g. sale of recovered resources, lower water treatment costs) and non-use values (e.g. averted health or time costs) is applied (Haller et al., 2007; Alam, 2008).

Major benefits that have been monetized and are associated with sanitation development are (1) health, (2) access time/welfare, (3) water and environmental quality, and (4) revenues from resource recovery (Winara et al., 2011; Hutton, 2013).

1.2.4.2 Need for an integrated approach to evaluate costs & benefits of sanitation

Individual cause-effects to evaluate costs and benefits of different wastewater and solid waste interventions have been well described in literature, such as (1) the effect of discharging a pollution load on the quality of receiving water (e.g. Hatt et al., (2004), Suharyanto & Matsushita, (2011)), (2) the effect of sanitation intervention on improvement of public health (e.g. Malekpour et al. (2013)), (3) impact of wastewater interventions on discharged pollution loads (e.g. Suwarno et al. (2013)), (4) economic losses as a result of poor sanitation (e.g. Hutton (2013)), and (5) the potential to recover resources from sanitation interventions (De Graaff et al., 2011; Cornejo et al., 2013; Tervahauta et al., 2013). However, no integrated framework exists in scientific literature that may fully evaluate the costs and benefits of sanitation improvements on a national scale.

1.3 Objective of this thesis

The objective of this these is to develop a framework that can quantitatively evaluate a set of alternative wastewater and solid waste systems and allows for resolving costs and benefits across spatial and temporal scales, and sustainability dimensions (social, environmental, economical) on a nationwide scale.

It is hypothesized that access to sanitation in developing countries and specifically Indonesia can be accelerated by an increased cost benefit ratio resulting from resource recovery.

To test this hypothesis, a sanitation planning framework will be developed and demonstrated, using Indonesia as an example. Following the identified knowledge gaps described in paragraph 1.2, it links the following four elements:

1. Analysis of the technical and financial feasibility of wastewater and solid systems in relation to the residential (urban/rural) features;
2. Development of a sanitation planning framework that translates targets and implementation into budgets allocated to responsible institutions for implementation. The impact of interventions on production of recoverable resources and consumption (e.g. energy, area) as well as (wastewater) pollution discharged should be made visible;
3. Spatial and temporal comparison of the potential demand and supply of recoverable resource from wastewater and solid waste;
4. Analysis of the Benefit to Costs Ratio (BCR): The final step involves the development of an approach that links the overall costs (OPEX + CAPEX) with the Benefits of wastewater and solid waste interventions (health, environment & water quality improvement, socio-economic, land values increase and sale of recoverable resources.

1.4 Background on current Indonesian wastewater and solid waste sector

The development of the framework is illustrated using Indonesia as an example. While access to improved sanitation facilities in South East Asia has reached 72%, Indonesia is lagging behind with only 61% having access (WHO & UNICEF, 2015). In addition, economic losses due to the absence of sanitation in Indonesia are nearly a factor 10 higher than those of other Asian countries, such as the Philippines, Cambodia and Vietnam (Hutton et al., 2008). Moreover, the government of Indonesia has committed itself to provide 100% access to sanitation before 2020 (Bappenas, 2014).

Located in Southeast Asia, The Republic of Indonesia is an archipelago bordering Australia, East-Timor, Malaysia, Papua New Guinea, the Philippines and Singapore (see Figure 1.3).

Figure 1.3 The Republic of Indonesia (Wikimedia, 2014)

In the coming 20 years Indonesia's population is expected to grow from the current 250 million to over 305 million people (BPS, 2013). An equally remarkable urbanization is forecasted, as by 2035 an estimated two third of the Indonesian population will live in urban areas compared to the current 50% (BPS, 2013). Nearly 60% of the population lives on Java (BPS, 2013).

The vast majority of households in Indonesia with access to wastewater facilities relies on septic tanks (WSP, 2013). A septic tank is the minimum treatment requirement in Indonesia (BPS, 2014) and the construction and maintenance is the responsibility of private households (WSP, 2011).

Despite the availability of design standards for septic tanks (MoPW, 2000), these are rarely enforced due to the absence of institutional capacity and, consequently, 95% of septic tanks leak and result in the pollution of groundwater (WSP, 2013). Community based systems or SANIMAS (Indonesian: *Sanitasi oleh Masyarakat*) comprising a community sanitation center or a simplified sewer system of small diameter pipes connected to an anaerobic baffled reactor, have been gaining grounds (Ulrich et al., 2009; Roma & Jeffrey, 2010; Reynaud et al., 2012b). By 2010, nearly 600 of such systems were implemented with 5,000 additional systems planned for the near future (Eales et al., 2013; Kearton et al., 2013). Evaluation of these systems (Reynaud et al., 2012a; Eales et al., 2013) confirmed the technical capabilities of the anaerobic systems to meet applicable effluent standards (MoE, 2003). However, challenges were identified such as the division of roles and responsibilities in technical and financial management, and the removal and safe disposal of sludge (Eales et al., 2013).
By 2012, only 12 centralized municipal wastewater treatment plants (WWTP) were in operation in Indonesia serving less than 1% of the population (USAID (United States Agency for International Development), 2006; Kearton et al., 2013). The systems utilized were (aerated)

lagoons, UASB (Upflow Anaerobic Sludge Blanket), Rotating Bio Contactors (RBC's) and activated sludge systems (Kearton et al., 2013). Poor sewer network quality causes seepage of groundwater into the network, which dilutes the sewage and increases the flow to the treatment works (USAID, 2006). Connecting households to the sewer systems is a major problem and requires institutional strengthening and advocacy (Whittington et al., 2000; Kearton et al., 2013; Winters et al., 2014). Several medium centralized WWT systems (serving 500 to 5,000 households), typically RBC's or Anaerobic Filters, were established in the past years (PDPAL-Banjarmasin, 2012) or are planned (Kearton et al., 2013).

Existing municipal solid waste (MSW) systems include the collection of waste from households by motorized or hand carts to a transfer station, followed by transportation to a landfill (TTPS, 2009; Aprilia et al., 2012). Between 2010 and 2014, 207 municipal landfills were constructed but only 132 have sufficient capacity until 2019 (MoPW, 2014). The government is aiming for a 20% reduction of (urban) waste landfilled through the promotion of the "3R concept (Reduce, Reuse, Recycle)" (Bappenas, 2011), which has resulted in the construction of approximately 300 communal 3R stations by 2014 (MoPW, 2013). A lively, but informal sector has developed that is active in the recovery of reusable solid waste components, such as plastics and paper (Sasaki & Araki, 2013; Chaerul et al., 2013).

The provision of sanitation services in Indonesia has been divided between central government for policy making and overviewing, and local governments for implementation following Law 32/2004 (ADB, 2013). However, the actual responsibilities for particular subsectors lie with individual line ministries and their corresponding offices at local level (WSP, 2014).

The Ministry of Public Works is the lead agency for providing sanitation infrastructure to urban and rural areas. It provides regulations for both wastewater systems (Decree 16/2008; covering (i) increased coverage of wastewater, (ii) increased community and private sector involvement , (iii) development of a regulatory framework for urban sanitation, (iv) capacity building for wastewater management, and (v) increased investment for wastewater infrastructure) and solid waste (Decree 21/2006; covering (1) waste reduction at the source, (2) participation of householders and local community organizations) (ADB, 2013). The Ministry of Health is responsible for behavior change with a strong focus on rural sanitation development. Under its leadership the Community-Led Total Sanitation (CLTS) (Mehta & Movik, 2010) Strategy has been implemented since 2005. The CLTS is based on 5 pillars, being hand washing with soap, hygiene and safe food and water treatment, safe wastewater management and solid waste management at household level. The Ministry of planning (BAPPENAS) is in charge of setting sector targets and policy development, whereas the Ministry of Home Affairs is responsible for capacity building for local governments (WSP, 2014).

1.5 Scope of this thesis

We first provide an analysis on how an integrated wastewater and solid waste management system may add value in terms of recoverable resources and financial benefits in a Chinese residential area development (**Chapter 2**). For years China has been undergoing rapid urbanization and economic development. However, these developments have negatively impacted the environment and China is suffering from severe water pollution as a result of discharge of untreated wastewater (Zhang et al., 2002; van Dijk & Mingshun, 2005). As a result, China plans to reach an 85% treatment target by 2015 and a 15% wastewater recycling rate by 2015 (Wang, 2012). In this analysis potential wastewater and solid waste treatment schemes applicable in the Chinese context are evaluated. Starting point is the closing of material cycles focusing on possibilities to recover and reuse valuable resources and energy from “waste” produced in an urban setting. Concepts with and with source separation will be compared.

Secondly we describe the core of Indonesia’s current wastewater improvement strategy (**Chapter 3**). Decentralized communal treatment systems are often promoted as the core of the sanitation improvement in Indonesia for their low cost, their decentralized features as well as their potential to effectively remove organic pollutants (COD, BOD) and solids (Ulrich et al., 2009). Since 2009 community-managed DEWATS (Decentralized Wastewater Treatment Systems) have been assigned a central role in reducing open defecation, improving urban sanitation and meeting its Millennium Development Goal for sanitation (Eales et al., 2013). Limited up to date information is available on the actual sustained performance of applied systems.

Taking the massive scale of planned implementation into consideration, an evaluation of the technical and financial-economic aspects and users’ involvement of implemented treatment systems is performed.

In Chapter 4, the technical and feasibility analysis of wastewater and solid waste systems in the context of Indonesia is presented. Based on the analysis, a principle, residential area-dependent, system selection is proposed that can be a direct input for a sanitation planning.

The development of a sanitation planning framework is described in **Chapter 5**. The principle selection criteria framework, developed in **Chapter 4**, is applied on the anticipated population and residential area development in Indonesia following the government’s sanitation access targets. The output comprise (1) visualization of implementation in GIS, (2) quantification of budgets and systems required, and (3) distribution of the budget according to implementing institutions

The developed approach to determine the potential demand for recoverable resources from wastewater and solid waste is presented in **Chapter 6**. Following past development, growth forecast and corresponding time bound and location dependent resource demand for agriculture (crops, plantations and horticulture), aquaculture and consumer goods (paper and plastics) are determined. The determined potential resource demand is finally compared to the potential recovery potential for Indonesia.

An integrated approach to evaluate costs and benefits of wastewater and solid waste management to improve water quality is introduced in **Chapter 7**. The functionality will be illustrated using the heavily polluted Citarum River in West Java (Indonesia) as an example. The presented approach will allow for a quantification of the impact of different types of interventions on (1) water quality improvement, (2) resource recovery potential, and (3) monetized benefits to costs ratio.

In **Chapter 8** the outcomes of these made steps are integrated and synthesized into a National Sanitation Planning (NaSaP) framework. The framework is discussed and an outreach to future extension possibilities is presented.

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Chapter 2

Designing Sustainable Sanitation in Urban Planning Proposed for Changzhou, China

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Abstract

China is undergoing rapid urbanization and economic development. This requires a new approach on spatial planning and environmental infrastructure. In the presented project an example of this approach is given for the city of Changzhou (China) where a new residential area (Qinglong district) will be developed for 100.000 inhabitants. Key issue within the formulation of sustainable sanitation concepts is the integration and management of water, waste and energy in such a way that they will become beneficial to the establishment of the envisaged green city. Starting point was the closing of material cycles focusing on possibilities to recover and reuse valuable resources and energy from “waste” produced in an urban setting. Four different scenarios focusing on water, nutrient and energy recovery were compared with the baseline wastewater management practice. Besides environmental benefits, the economic benefits of sustainable sanitation concepts are attractive, the break even point with the baseline scenario, is already after 5 years, provided that recovered resources will be sold for a marketable price.

We believe that presented concepts are applicable for a wide range of new urban development initiatives in China and similar rapidly developing densely populated regions worldwide.

Keywords: Sustainable sanitation, resource recovery, energy production, economic analysis, CO_2-emission reduction

2.1 Introduction

With annual growth rates frequently in the low-double digit range during the previous 15 years, China's economic development has been impressive. Various authors (Zhang et al., 2002; van Dijk & Mingshun, 2005) have identified a negative impact of this economic progress on the environment. China's already stressed environment is going through additional stress caused by the rapid industrialization and urbanization. Water scarcity and deteriorating water quality of rivers, lakes and groundwater are the result of industrial, municipal and agricultural sewage and drainage discharge (UNDP, 2005). Besides the low treatment rate, with a reported value of 34 % in 2002, also a low efficiency in water utilization and a high universal wasting of water are identified. Solving this is considered as the major challenge for China (Zhang et al., 2002).

The project described in this paper is located in Jiangsu province (South East China). Jiangsu province, as a coastal province, is one of five provinces that contributed a total of more than one third of China's GDP (Heilig, 2006). Changzhou city is located in the South of Jiangsu. The nearby located Taihu lake suffered from severe euthrophication in May 2007. As a result of this event and the expected population increase from today's 1.4 million to a planned 1.8 million people in 2020 and at the same time a planned increase of total GDP in the urban area of 138-150 billion RMB to 370-400 billion RMB in 2020, the Changzhou government will follow a sustainable economic development path. The Qinglong district is identified as a new area in which this sustainable development should be put into practice. Currently the area is used for some minor farming activities, but should, in 5 years from now, develop into an urban area housing 100,000 people. Qinglong is regarded to start from a greenfield situation.

2.2 Methods

Key issue for sanitation concepts is the integration and management of water, waste and energy. Applying this approach will help the government in achieving its sustainable economic ambition. In the presented concepts the focus is on closing material and energy cycles. Thus, "waste" is no longer perceived as a problem, but as a valuable resource that can be recovered. Optimizing the use of energy combined with the recovery of energy in waste streams contributes to China's national ambition of reducing CO_2 emissions.

The possibilities and virtues for resource recovery from wastewater and organic waste have been extensively investigated (Otterpohl, 2001; Lettinga, 2006). With the different resource recovery possibilities a distinction has been made between three major opportunities:

1. Reuse of high quality effluent produced with Membrane Bioreactor (MBR) technology (Van Bentem et al., 2006);
2. Recovery of energy through application of anaerobic treatment (Kujawa-Roeleveld et al., 2005);

3. Recovery of nutrients through application of struvite precipitation (MAP process) focusing on P-recovery (Maurer et al., 2006);

In present study enhanced N-recovery processes like electrodialysis, reverse osmosis or ion exchange were not included, as these are not successfully proven yet (Van Voorthuizen et al., 2008).

2.2.1 Studied scenarios

Based on the identified potential usage of these recovered resources, four scenarios were developed employing source separation and were compared to a baseline scenario (Figure 2.1).

2.2.2 Wastewater characteristics

Calculations of greywater and black water and urine parameters were based on Kujawa-Roeleveld, (2005). Table 2.1 shows the applied values of the in-house water consumption.

Table 2.1 Applied values for in-house water consumption [a, b]

Parameter	Unit	Value
Greywater	l/cap/d	90
frequency female WC faeces + urine	1/d	1.5
frequency female WC urine	1/d	6
frequency male WC faeces + urine	1/d	2
frequency male WC urine	1/d	2
frequency male urinoir	1/d	4
water use WC male/female to blackwater	l/flush	6
water use WC male/female to urine	l/flush	3
water use urinoir	l/flush	0.1

[a] In China separate discharge of rainwater from wastewater is common practice.

[b] For the irrigation of city parks a value as presented by the city council of 0,75 m^3/m^2/year is applied.

2.2.3 Efficiency of treatment units and conversion factors

An anaerobic COD removal efficiency of 70% is applied for blackwater treatment (Elmitwalli et al., 2006; De Mes, 2007). Effluent characteristics and energy data for an MBR were obtained from the full-scale practical MBR in Varsseveld (Van Bentem et al., 2006). Nitrification and denitrification are assumed to proceed via conventional processes. Calculations of CO_2 emission were based on oxidation of organic matter with an average TOC/COD ratio of 0.33 (Van Bakergem & Groen, 1998). For the calculation of CO_2 emission as a result of electricity production a CEF/MWh of 0.7822 is applied (http://cdm.ccchina.gov.cn).

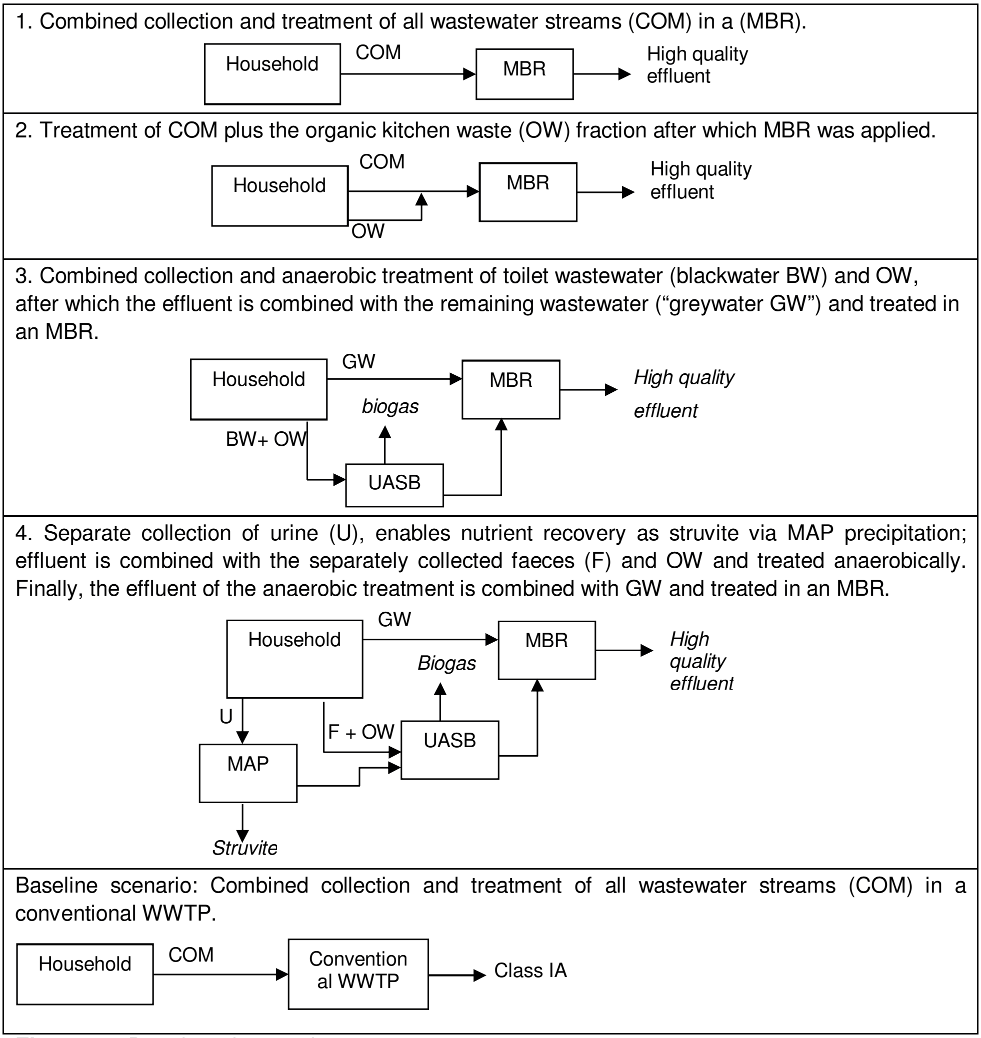

Figure 2.1 Developed scenarios

2.2.4 Economic costs and benefits

In 2008 10 RMB equaled ±1 euro. Costs for the sewer system were obtained via DHVs' local design institute and are 650 RMB/m for pipes inside the city (total 30 km) and 3.000 RMB/m for main pipes (additional 5 km in baseline scenario). For separate collection of blackwater and greywater, costs were 1.5 times higher and 1.7 times higher in case of additional separate urine collection (communication with the DHV sewerage department).

Presented capital costs (CAPEX) were based on DHV's estimates of realized projects and converted to the local Chinese situation by the DHV China office and were calculated by annuities in which the depreciation period for civil works and the sewer system is 30 years, whereas for mechanical and electromechanical equipment 20 years was applied. Operational (OPEX) costs (sludge treatment, labour, maintenance) as well as Consumer Price Index (8.3%), energy price increase (9.8%) and interest (7.8%) were obtained via Chinese local sources. Membranes need to be replaced every 5 years and are considered part of OPEX. Determination of the running costs over time was based on the sum of indexed OPEX and CAPEX. All assumed values in this paper, e.g. sewer price, price for the products, energy price etc., were predicted based on historical market development but can be subject to change. A sensitivity analyses can help to identify the influence of a change of each individual value on the total costs. However, at this stage we consider this outside the scope of this paper.

2.3 Results and discussion

2.3.1 Wastewater production

The characteristics of produced wastewater streams in different scenarios are presented in Table 2.2.

Table 2.2 Wastewater characteristics in the scenarios; COM: all wastewater streams OW: organic kitchen waste, BW: blackwater, GW: greywater, F: faeces, U: urine

Parameter	Unit	Scenario 1	Scenario 2	Scenario 3/4	Scenario 3	Scenario 4	
		COM	COM+ OW	GW	BW+ OW	F+OW	U
Flow	m^3/d	11,260	11,315	9,000	2,315	1,100	1,215
COD	mg/l	1000	1520	445	5700	11070	800
TN	mg/l	85	105	11	470	500	440
TP	mg/l	17	25	4	106	152	62

2.3.2 Energy data

Figure 2.2 shows the energy data for the operation of the four different scenarios. In the baseline scenario surface aeration is applied, as this is currently widely applied in China due to its robustness and low maintenance. In the other presented scenarios bubble aeration is applied, as this is generally more energy efficient.

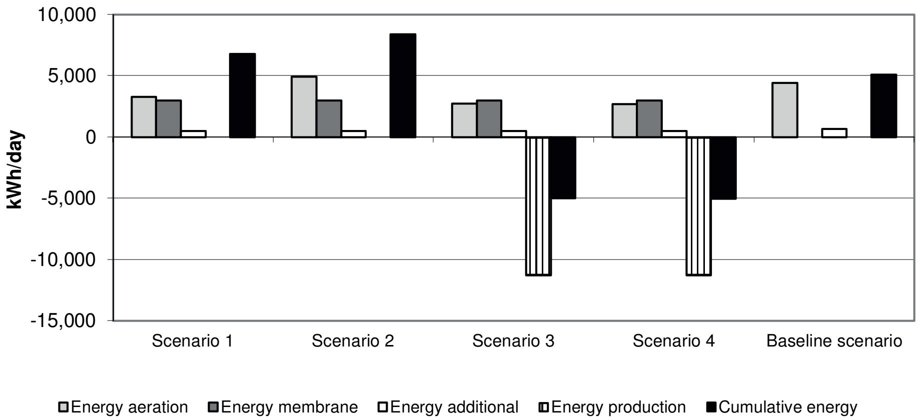

Figure 2.2 Energy produced and consumed in the treatment process

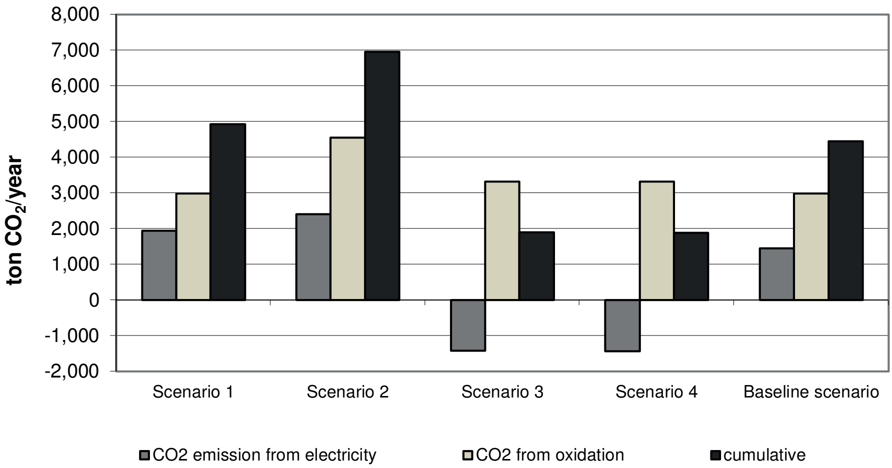

Figure 2.3 CO_2 emissions in the different scenarios

Compared to the baseline scenario, application of an MBR (Scenario 1) consumes more energy. Based on the loading, only Scenario 2-4 can be compared, since here the organic fraction of the kitchen waste is included. From that perspective, it becomes clear that Scenario 2 is worst in terms of energy consumption, due to a high energy input requirement for aeration as well as for membrane filtration. Both Scenario 3 and 4 result in an energy producing system, despite the application of an MBR. Scenario 4 is most favorable as part of the nitrogen has been removed during MAP precipitation process and therefore less nitrification and thus aeration is required.

2.3.3 CO_2 emission

In a WWTP the total CO_2 emission is the sum of direct and indirect processes. Direct CO_2 emission is the result of oxidation of organic pollutants. Indirect CO_2 emission is due to the production or consumption of electricity. Figure 2.3 shows CO_2 emission from both processes for the four scenarios and baseline scenario. CO_2 emissions related to the electricity requirements are in line with the energy data. However, Scenarios 3 and 4, in which part of the organic pollutions is converted anaerobically, thereby producing a far smaller amount of CO_2 show a CO_2 emission reduction as well. The yearly CO_2 emission reduction between Scenario 3&4 and Scenario 2 is about 5,000 tons. Compared to the baseline scenario a 2,500 tons reduction is achieved. However, part of the produced methane, which is considered to be a 21 fold stronger greenhouse gas than CO_2, will dissolve in the effluent of the anaerobic step in Scenarios 3 and 4 and be emitted to the atmosphere in the MBR after all. According to Van Haandel & Lettinga (1994) the dissolved methane represents under equilibrium conditions 64 mg/l as COD. Thus, in Scenario 3 and 4 will emit, as a worst case scenario, an additional CO_2 of 280 ton CO_2/y, which equals about 10% of the achieved reduction. Today's Clean Development Mechanism in which a price for each ton of CO_2 of € 5 is applicable, provides an additional driver for implementation of sustainable sanitation concepts in which anaerobic treatment is included. In the remaining cost benefit analysis this additional income is not included.

2.3.4 Water saving

In the governments' planning it is indicated that one person consumes between 140-160 l/cap/d. In this project the starting point was that the daily water consumption should not be limited, but the quality does not necessarily need to be the same as of municipal tapwater. It must be noted that the provided amount is based on water used for in-house application (drinking, toilet, showering, etc) as well as external community oriented water use (e.g. road cleaning and gardening). In the present project the daily (in-house) water consumption is determined as 113 l/cap/d of which 90 l is greywater (Table 2.1). In contrast to the baseline WWTP, an MBR (Derksen et al., 2006; Van Bentem et al., 2006) will comply with the Chinese standards for different uses such as toilet flushing or gardening. A conventional WWTP can in general comply with effluent quality class IB and IA, and irrigation for some non-food agricultural purposes. Table 2.3 shows the standards, which are referred to.

Table 2.3 Discharge limits primary pollutants Class IA and B GB18918-2002 and Control Indexes of Reclaimed Water for Miscellaneous Use (GB/T18920-2002)

Parameter	Unit	Discharge China GB18918-2002		"Miscellaneous" use of reclaimed water China GB/T 18920-2002				
		IA	IB	Toilet flushing	Road cleaning & fire fighting	Gardening	Vehicle wash	Civil constr.
COD	mg/l	50	60	-				
BOD_5	mg/l	10	20	10	15	20	10	20
SS	mg/l	10	20	-				
NH_4^+-N	mg/l	5 (8)*	8 (15)*	10	10	20	10	20
TN	mg/l	15	20					
TP	mg/l	0.5	1					
Turbidity	NTU			5	10	10	5	20
TDS	mg/l			1500	1500	1000	1000	-
DO	mg/l			1.0				
pH		6.0-9.0		6.0 - 9.0				
Fecal coliform	Units/l	10^3	10^4	3				

* Figures out of brackets are for water temperature > 12ºC, whereas the figures in brackets are requirement for water temperature ≤ 12ºC

Therefore in all four scenarios the effluent of the MBR can be reused for all aesthetic related applications as well as for toilet flushing. All other in-house water applications will still be fed by municipal tapwater. Applying the aforementioned, the daily total water consumption is 157 l/cap/day of which only 90.5 l/cap/d is required of municipal tapwater, which equals over a 40% reduction.

2.3.5 Struvite production

Scenario 4 includes the struvite production from urine of which an amount of 220 ton/year of (hydrogenated) struvite is expected. In order to realize a source-separate urine collection system, a more complex and elaborated sewer system is required. It is acknowledged that alternative methods for urine collection are available as well (Mels et al., 2005). Additionally a reactor aimed for struvite precipitation is required. Based on Shu et al. (2006) struvite can be sold at a value of 460 Euro/ton (~4,600 RMB/ton). This price level might not be realistic at this moment (local information), but increasing scarcity of phosphate ore (www.fullermoney.com, 2008) will ultimately have its effect on struvite selling prices.

2.3.6 Yearly operation costs.

Figure 2.4 shows the CAPEX based on annuities for the establishment of the presented WWTP and the required sewer system. Specific yearly CAPEX for sewer system ranges from

17 RMB/cap in Scenario 1 and 2 up to 25, 29 and 30 RMB/cap for, respectively, Scenarios 3 and 4 and the baseline scenario. For the WWTP the specific yearly CAPEX values are 58, 66, 81, 85 and 29 RMB/cap for, respectively, Scenarios 1-4 and the baseline scenario. It is found that establishment of Scenarios 3 and 4 involve almost double the costs of the establishment of the baseline scenario.

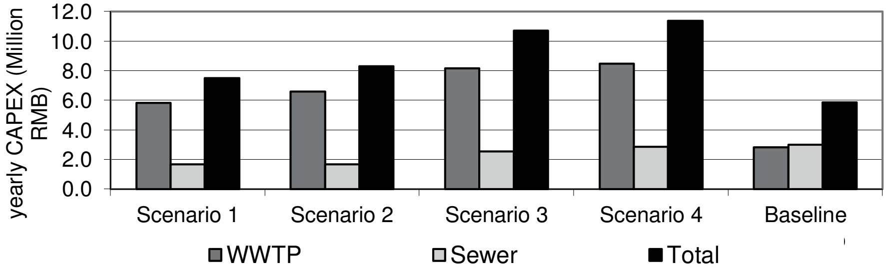

Figure 2.4 CAPEX for the establishment of the full WWTP infrastructure (million RMB)

Analysis of the OPEX (Figure 2.5) shows that Scenario 3 and 4 (both energy producing) are most cost effective, followed by Scenario 1. Scenario 2 comes out less favorable than the baseline scenario. In this estimation it is assumed that 40% of the water that is saved (which equals the amount of water required for toilet flushing and gardening) is sold at the price of 2.4 RMB/m^3, whereas an additional (assumed) 30% is sold for agricultural purposes at 0.5 RMB/m^3. This additional 30% is also sold at the same price in the baseline scenario.

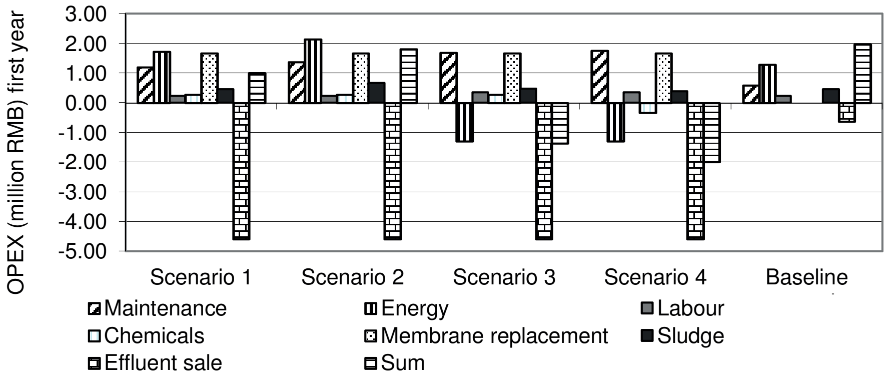

Figure 2.5 OPEX of the presented scenarios

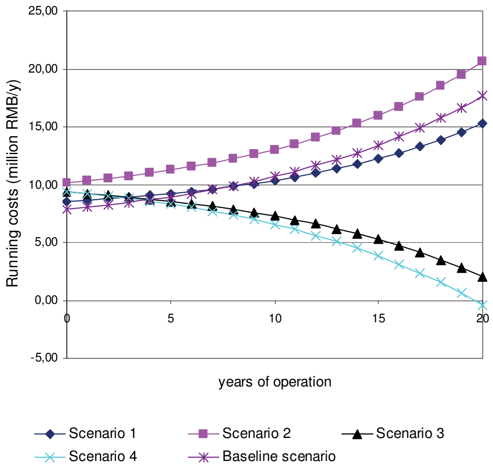

Figure 2.6 Development of running (CAPEX and OPEX) costs

When looking at the development of total yearly running costs in time (Figure 2.6) it appears that Scenario 2 is least favourable, followed by Scenario 1 and the baseline scenario. Only Scenario 3 and 4 show a distinct pattern, as they, despite high initial investments, result in decreasing running costs. Break-even point with the baseline scenario is after 5 years. Because of the additional struvite production Scenario 4 is the most favorable. Again it is emphasized that the treated organic load in Scenario 1 and baseline scenario are smaller than in Scenario 2-4. Therefore, the former would require an additional infrastructure for the collection, transfer and treatment of organic kitchen waste.

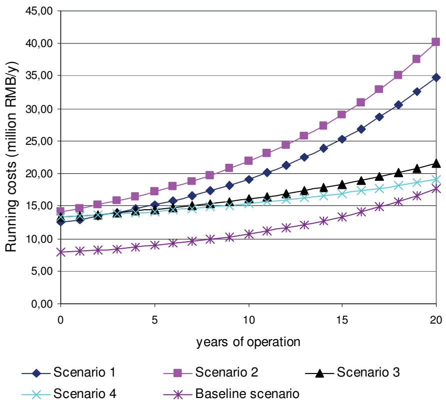

Figure 2.7 No MBR effluent sale for high quality purposes

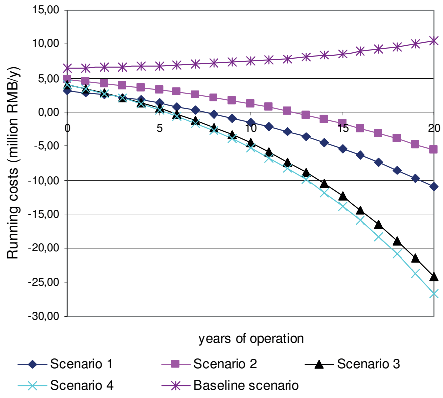

Figure 2.8 All MBR effluent sold for high quality purposes

2.3.7 Risk & Sensitivity

- In the presented scenario the starting point is that all water for the toilet flushing and park irrigation is obtained from treated effluent. Figure 2.7 and Figure 2.8 show the results when no effluent is sold for higher quality effluent purposes (still 30% used for low-quality purposes) and all effluent from MBR is sold as higher quality effluent (and 100% of the baseline scenario as low quality). Thus, it becomes clear that the application of an MBR only becomes economically attractive if a considerable amount of effluent can be sold for higher quality effluent purposes;
- The effect of small footprint, which greatly increases the economic attractiveness of MBR application, was not included. In a fast urbanizing country like China, this effect will increase;
- It must be taken into consideration that source separated systems bring in a risk of misconnection of the different sewer lines, as was shown in the case of Leidsche Rijn (www.waterforum.net; 20-06-2002) Experiences (personal communication with A. During) reveal that about 1% is falsely connected immediately after finalizing the infrastructure, whereas this number can increase to 15% over time due to private/illegal interventions;
- Different scenarios and transport means are available to convey/collect blackwater to a treatment location. In case of a decentralized treatment step, application of vacuum toilet has been successfully demonstrated (Kujawa-Roeleveld et al., 2005; Meulman et al., 2008). In present scenarios it is assumed that for the 100,000 people only one central treatment plant is constructed for which a sewer system based on vacuum toilet is not found feasible due to the long transportation distances and problems with clogging. When a water saving toilet is applied without the mixing of greywater such problems are still a possibility, which can be overcome when transportation is realized via a pressurized system. The required pump energy for this transport was excluded;
- Studies have been performed in which the acceptation of source separated systems has been identified as a potential problem for successful introduction (Lienert & Larsen, 2006). This issue has not been extensively incorporated in present study;
- By offering the possibility for extensive effluent reuse for non-agricultural purposes (park greening), as proposed in the four scenarios, the aesthetics of the environment will be improved. Previous projects in which an increase of green in the living environment was achieved resulted in an increased “willingness to pay” for users of such areas. In this study this price increase has not been included.

2.4 Conclusions

The outcomes of the study that were performed for the Changzhou project show that:

- By introducing an alternative sanitation several environmental benefits can be realized, resulting in a 40% water saving (by closing part of the water cycle), production of energy (by application of anaerobic technology), reduction of CO_2 emission, and struvite recovery (by treating separate collected urine);
- Economic analysis shows that application of a source separated sewer system followed by treatment in an anaerobic and struvite precipitation and post treatment in an MBR becomes already attractive after a period of 5 years, provided the recovered resources (water, energy and nutrients) can be sold at mentioned prices. The presence of long term and reliable buyers are essential for the successful economic applicability of presented scenarios;
- Besides the realization of the two basic goals for sanitation (assuring the public health and preventing the environmental pollution), this paper shows that by introducing sustainable sanitation concepts an enhanced resource recovery is possible which stimulates the economic development of rapidly emerging urban areas in China and similar regions.

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Chapter 3

Evaluation of DEWATS in Java, Indonesia

This chapter has been published in a slightly modified version as:

Evaluation of DEWATS in Java, Indonesia

Kerstens, S.M., Legowo, H.B. & Hendra Gupta, I.B. (2012*), Journal of Water, Sanitation and Hygiene for Development* , Volume 2, Issue 4: 254-265, DOI 10.2166/washdev.2012.065

Abstract

Under the Indonesian PPSP (Accelerated Sanitation Development for Human Settlements Program) thousands of new DEWATS (Decentralized Wastewater Treatment Systems) may be realized in the coming five years. Taking the massive scale of planned implementation into consideration an evaluation of the technical and financial-economic aspects and users' involvement for three different types of DEWATS was performed. Evaluated systems included (1) Settler (Set) + Anaerobic Baffled Reactor (ABR) + Anaerobic Filter (AF), (2) Digester + Set+ ABR + AF and (3) Settler, equalization, activated sludge, clarifier, filtration. All three systems complied with the current regulations. System 3 suggested the best overall performance on selected parameters in the monitored period. A clear reduction in specific investment costs per household was found with an increasing number of households per system. Only daily, regular operational costs were recovered from fees collected by the community, whereas costs for desludging, major repairs and capital and replacement costs were not. Surveys with users showed a different degree of involvement of local men and women in the planning stages of the project for the systems. Recommendations are provided to scale up the introduction of DEWATS in a more sustainable way in the framework of a city wide sanitation strategy.

Keywords: DEWATS, financial, Indonesia, performance evaluation, sanitation, technical

3.1 Introduction

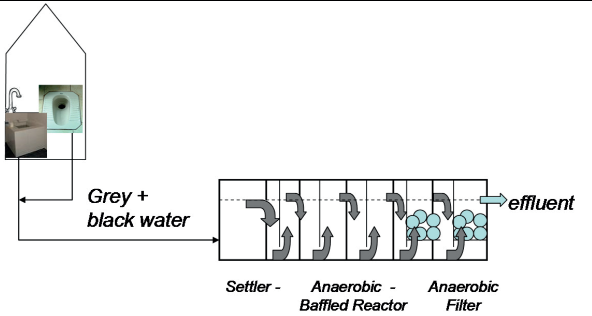
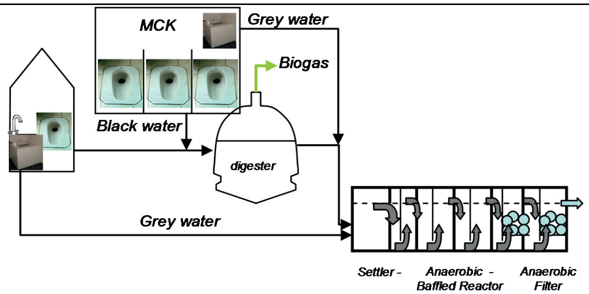
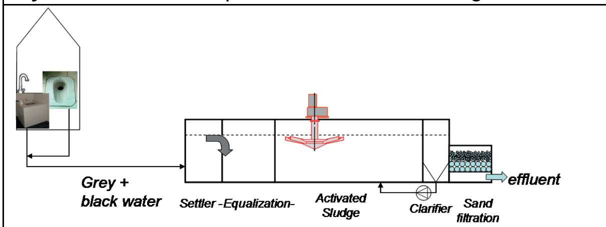
Access to improved sanitation in Indonesia is below most other South East Asian countries, with an approximate 80 million people still using open defecation. The Water and Sanitation Program's (WSP) Economic Impacts of Sanitation in Indonesia (Napitupulu & Hutton, 2008) estimated the overall economic losses from poor sanitation at approximately US$ 6.5 x 10^9 annually. Decentralized communal treatment systems are often promoted as the core of the sanitation improvement in Indonesia for their low cost, their decentralized features as well as their potential to effectively remove organic components and solids (Ulrich et al., 2009). Limited up to date information is available on the actual sustained performance of applied systems. Mostly, the effluent of DEWATS (Decentralized Wastewater Treatment Systems) is discharged locally. Evaluation of the effluent is important with respect to safeguarding public health and the environment (Vollaard et al., 2005), in particular when the body receiving the effluent waters is having particular functions for which it must meet related water quality criteria. In addition, this paper compares the financial and economic aspects of the different systems as well the level of involvement of the users. Analysis from these three perspectives (technical, financial-economic and social) is considered essential for improved and accelerated access to safe sanitation, upkeep with population growth and sustained and equitable service delivery in Indonesia.

3.2 Methodology

3.2.1 Site selection

In consultation with the Ministry of Public Works (MoPW), three municipalities in Java were selected for evaluation. These were Yogyakarta, Surakarta (also known as Solo) and Blitar. In each of the municipalities three DEWATS were selected. Site visits were facilitated by local partner LPTP (Lembaga Pengembangan Teknologi Pedesaan or Institute for Rural Technology Development) in Yogyakarta and Surakarta and by the local bureau of environment in Blitar. Three types of systems were evaluated. The first two types are the commonly applied systems, whereas the third type seems to have been introduced only recently, but is gaining in popularity in the Blitar area. Because of its electricity use, this third type may not be classified as a DEWATS by all stakeholders (Ulrich et al., 2009).Table 3.1 shows the key features of the nine evaluated sites.

Table 3.1 Characteristics of evaluated DEWATS project sites organised by system

<table>
<tr><td>System 1: Settler + Anaerobic Baffled Reactor (ABR) + Anaerobic Filter (AF)</td><td>Project name, province, Kota/ Kabupaten (year); number of households (hh)</td></tr>
<tr><td rowspan="5">
</td><td>1. Minomartani, DYI, Yogyakarta, Kab. Sleman (2007); 70 hh.</td></tr>
<tr><td>2. Santan, DYI, Yogyakarta, Kota Yogyakarta (2010); 80 hh</td></tr>
<tr><td>3. Kragilan, Central Java, Surakarta, Kota Surakarta (2005); 100 hh</td></tr>
<tr><td>4. Sukorejo, East Java, Blitar, Kota Blitar (2003); 200 hh</td></tr>
<tr><td>5. Karang wetan East Java, Blitar, Kota Blitar (2006); 100 hh</td></tr>
<tr><td colspan="2">System 2: Separate black (BW) and grey water (GW) collection. BW of households (and MCK[1]) to digester; effluent of digester with GW to Settler + ABR + AF</td></tr>
<tr><td rowspan="3">
</td><td>6. Gambiran, DYI, Yogyakarta, Kab. Sleman (2008); no MCK [a] applied; 50 hh</td></tr>
<tr><td>7. Pajang, Central Java, Surakarta, Kota Surakarta (2010); including MCK; 40 hh connected; 29 hh use MCK</td></tr>
<tr><td>8. Srenang, Central Java, Surakarta, Kota Surakarta (2008) including MCK; 22 hh connected; 44 hh use MCK</td></tr>
<tr><td colspan="2">System 3: Settler, equalization activated sludge, clarifier, filtration</td></tr>
<tr><td>
</td><td>9. Kepanjen Kidul East Java, Blitar, Kota Blitar (2010); design capacity 400 hh (only 100 hh connected at time of evaluation)</td></tr>
</table>

[a] MCK: *Mandi Cuci Kakus*: Communal toilet and washing facility

3.2.2 Laboratory analysis

For each site four samples were taken with the following schedule: Yogyakarta: 8, 17, 22 and 25 February 2011; Surakarta: 9, 18, 23 and 28 February 2011; Blitar: 16, 21, 24, and 28 February 2011. February is the rainy season. All evaluated systems apply a separate rainwater collection. However, some rainwater intrusion may have taken place. All samples were analyzed on the following parameters, with corresponding methods: pH (SNI 06-6989, 11-2004), Biological Oxygen Demand (BOD) (APHA, 2005, Section 5210-B, Section 4500-OG), Chemical Oxygen Demand (COD) (APHA, 2005, Section 5220-C), Total Suspended Solids (TSS) (in house method, spectrophotometer HACH, DR 2010), Total N (in house method, titrimetri), NH_4-N (SNI 06-2479-

2004), PO4-P (APHA, 2005, Section 4500-PD). All samples were collected and analyzed by the same laboratory (Balai Teknik Kesehatan Lingkungan (BTKL) in Yogyakarta). Influent samples were not taken because variations in quality and quantity are generally very high for small communities. Further, because all communities concern domestic users only, a comparable influent is expected.

3.2.3 Surveys and questionnaires

For each site two types of surveys were conducted. In a first survey representatives (all male) of each of the KSM (Kelompok Swadaya Masyarakat; "Community Independent Group") were interviewed on technical, institutional and financial features of the sanitation facilities. In the second survey six local surveyors interviewed a total of 90 respondents (59% females and 41% males) at 9 sampled locations, corresponding with 10 households per site. The topics were their involvement and satisfaction as users and tariff payers with the service delivery and service management (including financial management). All respondents were randomly selected and were part of the connected users in every DEWATS site. The age of the respondents involved in the survey was 28 years and older. Most of them were female (59%). Of these respondents, 11% did not have any formal schooling, 27% were elementary school graduates, 19% junior high school graduates, 30% high school graduates, and 13% university graduates. Most of the respondents' monthly incomes ranged between US$50 and US$100 (49%), 28% of the respondents had an income below US$50, 17% of the respondents had incomes between US$100-US$200 and 7% of the respondents' incomes were above US$200. ninety-two percent of the respondents had never been involved in a survey before.

3.2.4 Financial and economic analysis

For each of the sites information on the capital and operational costs as well as the nature and level of the user's fees were obtained during the discussion with the KSM and were – for reference – compared with the design documents. Construction costs included costs for the treatment plant itself, piping, digester (if applicable), MCK (if applicable) and facilitation (including training and project planning by BORDA "Bremen Overseas Research and Development Association" and LPTP or the local bureau of environment in Blitar). Taxes have been excluded from the presented data. To allow for comparison all investment costs were converted to the year 2010 price level, using the inflation percentage provided by the World Bank (http://data.worldbank.org/) and the CPI (Consumer Price Index) provided by Badan Pusat Statistik (http://dds.bps.go.id/eng/). To calculate the annual Capital Expenditures (CAPEX), investment costs were multiplied with an annuity factor based on a depreciation period of 20 years and an interest rate of 7%, which can be considered as a typical rate for Indonesia (www.web.worldbank). Finally, the economic losses due to poor sanitation were calculated based on WSP (Napitupulu & Hutton, 2008) and were corrected with the same CPI as mentioned above. The size of the population of Indonesia in 2010 was based on the 2010 Census and set at 238 million people. It is assumed that the household incomes obtained from the surveys are representative for the total community. In this study 1 US$ is equal to 9,000 IDR.

3.3 Results and discussion

3.3.1 Technical evaluation

Table 3.2 shows the effluent values of the three evaluated sites after four sampling rounds as well as the applicable effluent standard (MoE, 2003). System 1 is based on five sites, the values of System 2 on three sites and the values of System 3 on one site only (see also Table 3.1).

Table 3.2 Average and standard deviation effluent values and effluent standard [a]

Parameter	Unit	System 1: Set-ABR-AF	System 2: Dig-Set-ABR-AF	System 3: Act Sludge-Filt	Standard (No. 112, 2003)
pH	-	6.9 (0.3)1	7.2 (0.3)	7.2 (0.5)	6-9
BOD	mg/l	49.7 (8.2)	50.0 (15.7)	29.9 (11.9)	100
COD	mg/l	121.8 (21.9)	131.1 (53.1)	79.5 (39.5)	
TSS	mg/l	41.7 (24.5)	43.6 (30.0)	21.8 (10.3)	100
Total-N	mg/l	77.1(23.5)	88.0 (25.0)	58.7 (12.0)	
NH_4-N	mg/l	46.0 (20.6)	57.4 (26.7)	34.7(18.8)	
PO_4-P	mg/l	3.8 (1.8)	4.8 (1.8)	3.0 (1.9)	

[a] Values in brackets are standard deviations.

3.3.1.1 Organic pollutants

All three types comply with the 2003 regulations on pH, BOD and TSS. However, current effluent requirements are not very stringent, especially in comparison with applicable standards in neighboring countries. Philippines and Malaysia apply BOD effluent requirements of 30-80 mg/l and 20-50 BOD mg/l respectively (DENR, 1990; MDE, 2000). More stringent effluent requirements and extension on number of parameters (e.g. nutrients) should be considered in due course following, for example, a similar approach as the progressively increased raised standards in Malaysia to improve, in order of importance, public health, water quality and environmental quality. The data suggest that System 3, involving activated sludge and filtration performs significantly better on all effluent parameters compared to the other two system types, although it must be noted that System 3 is currently only operated at a quarter of its design load. The effluent values of System 1 (Settler + ABR + AF) and System 2 (Digester + Settler + ABR + AF), however, are comparable. Taking the high standard deviations of all systems into consideration it appears that stable operation is hard to achieve. This variation is also observed in the individual sites. A pilot study (Dama et al., 2002) on the ABR showed COD effluent values ranging between 50 mg/l and 400 mg/l with steady state values of typically 200 mg/l. Current measured values on the two types of systems applying ABR are considerably lower than these values. Unlike in Dama's study the currently evaluated systems have an additional anaerobic filter step, indicating that this filter contributes significantly to a better COD effluent. Wibisono et al. (2003) showed effluent COD values ranging from 80 to 144 mg/l COD in three piloted low-costs anaerobic systems in Indonesia, which are comparable to the values in the current study.

3.3.1.2 Nutrients (N & P)

The lower NH_4-N values in System 3 as compared to the other two systems are attributed to nitrification in the activated sludge system. In addition, the lower total N values in System 3 are attributed to denitrification. System 3 was only operational for one month and nitrification capacity is expected to increase further. At the same time, because the system is under loaded the results may differ compared to treatment of the full design load. Although the effluent P-concentration in System 3 is lower than in System 1 and 2, 3 mg/l, 4.8 mg/l and 3.8 mg/l respectively, the values are more comparable to each other than the other parameters. In general P is only removed to a limited extent in an anaerobic system. In aerobic systems, P-removal would require alternating anaerobic and aerobic conditions or the addition of iron or aluminum salts (Brett et al., 1997), none of which are applied. The lower effluent P-value in the aerobic system is attributed to the higher P-uptake by aerobic biomass compared to anaerobic biomass. The presence of P is the most common cause of eutrophication in fresh waters, with P-concentrations in streams of 20 µg/l already becoming problematic. Especially for smaller streams with multiple sources, such values are likely to be exceeded easily. In addition, with N/P ratios exceeding 16 (which is the case in all three systems) nitrogen also becomes a stimulant for eutrophication (Correll, 1998).

3.3.1.3 Coliforms

Levels of fecal coliforms discharged by activated sludge systems with filters (System 3) can be up to 1 or 2 log lower than levels in anaerobic filter systems (Systems 1 and 2) (Tchobanoglous et al., 2003). Currently no maximum discharge limit is formulated in the applicable 2003 law for these parameters. The World Health Organization (WHO) has defined several standards for reuse (WHO 2006) in agriculture or aquaculture, which are exceeded even for System 3. The Indonesian drinking water regulation for 2010 (number 492) requires the complete removal of all coliforms. Several KSMs reported the perceived decrease of diarrhoeal diseases since the establishment of the DEWATS service, but no supportive actual clinical data could be provided.

Reasons mentioned for the perceived decrease were both the reduced contamination of groundwater by previously applied and now replaced soakage / pit latrines and the reduced contamination of surface water with which women, children (especially boys) and men have direct contact during clothes washing, bathing and swimming. The quality of the discharged water is important from an environmental, economic and health perspective. Vollaard et al. (2005)

concluded in their study of the supply and bacteriological quality of drinking water and sanitation in Jakarta that inadequate disposal of human excreta is a threat to both piped water and groundwater. Phanuwan et al. (2006) showed the positive correlation between total coliforms and enteric viruses present in surface waters, showing the threat to protect public health from viral waterborne diseases. In addition, Charles et al. (2003) reported the increased discharge of Total Nitrogen (TN), Cryptosporidium and enteric viruses for small scale wastewater treatment systems (compared to a centralized wastewater treatment plant (WWTP)) as a result of insufficient operation and maintenance knowledge for these small scale systems. Thus, as a recommendation more attention to removal of pathogens could be paid in the design of DEWATS and their operation, as these pathogens have a direct impact on public health. Several low-cost post-treatment systems can be considered that have known effective removal of pathogens and nutrients, such as constructed wetlands or algae ponds (Laxton, 2010; Zhai et al., 2011). Mandatory disinfection to acceptable levels (WHO, 2006) could also be considered if reuse is strived for.

3.3.1.4 Feedback from operators and users

Discussion with system operators and the users revealed several issues that require attention:

- Systems are hardly desludged, even after years of operation. However scum formation in the settling compartment was grave and removal was required frequently (typically twice per month);
- Scum formation could be the result of attachment of biogas to incoming particles as described by Halalsheh et al. (2005). More frequent desludging could prevent this problem. This requires including the cost of sanitary desludging in the fees;
- Often, the removed scum and sludge were disposed of in the receiving water body. Besides the obvious contradiction of this practice to the objective of having a sanitation system, this approach is also a loss of potential safe fertilizer (Fach & Fuchs, 2010). Mostly provisions to deal with this differently were lacking as sites were not reachable by vacuum trucks or costs for frequent scum removal in a hygienic way were found too expensive. This aspect requires more attention in the designs, promotion and training;
- System 3 is identified as the most vulnerable one during the operation and maintenance (O&M) phase, as more skilled labour and understanding of the system and its electro-mechanical equipment as well as continuous electricity supply are required to operate the system in a sustainable manner;
- Manholes for both the network as well as the treatment system were often jammed. This limits the possibilities to operate and maintain the systems properly;
- During rainfall several systems become odorous. Although systems are designed to separate rainwater from wastewater, it was observed that rainwater can penetrate the manholes, which can be prevented by slightly elevating the manhole lids in the design. Odour problems are

probably caused by a reduced hydraulic retention time resulting in the discharge of not yet degraded organic (volatile) components;

- None of the systems was provided with an easily accessible sampling point (e.g. a sampling tap). It is suggested to include such a tap in new systems to facilitate sampling and monitoring the impact on the public health and environment. Further, it was mentioned that regular monitoring of effluent was not done or only in limited cases by either the responsible institutions or design or facilitation partners. If measurements were done, no feedback was provided to the community. More frequent monitoring and feedback to the community is recommended as this contributes to a better understanding of impacts and system performance;
- Finally, the KSM mentioned that in the design of the DEWATS, a citywide sanitation strategy was not taken into consideration, although mainstreaming of DEWATS is found to be a key element for sustainable infrastructure development (Ulrich et al., 2009). It is recommended to consider the application of a citywide sanitation strategy in the realization of DEWATS to both stimulate the introduction of DEWATS in systems where this is found feasible and, at the same time, limit introduction, where this is found less feasible.

3.3.2 Financial-economical evaluation

3.3.2.1 Investment costs

Figure 3.1 shows that the specific investment costs per household decrease as an increasing amount of households connected to the system. This “economy of scale” is seen both within one type of system (System 1 and 2) as well as for bigger systems in general. For System 1 (Set-ABR-AF) it was thus found that investment costs per household of a system for 200 households (US$ 240/household) are only approximately half of that of a system for 100 households (over US$ 460-490/household). This trend is not only found in the complete design (including piping, WWTP, facilitation, toilet blocks), but also in the specific investment costs per household for the piping system (Figure 3.2) as well as the treatment facility (Figure 3.3). The specific costs for the piping system for systems of type 2 (applying separation of black water and grey water) are considerably higher, because a double piping system is required. This is even more pronounced in the cases that combine individual house connections with an MCK (Srenang and Pajang), since in those cases the piping costs only reflect the connected households and not those that use the MCK. Based on these findings, it is recommended to look for opportunities to increase the scale of a treatment facility to reduce the investment costs. This could for example be achieved by having a joint DEWATS with two or more adjacent communities.

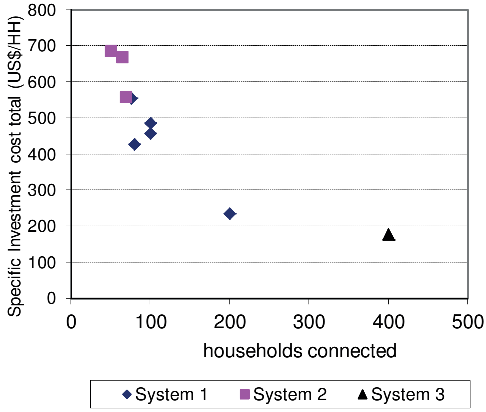


Figure 3.1 Specific total investment costs (US$) per household using the system, arranged by system

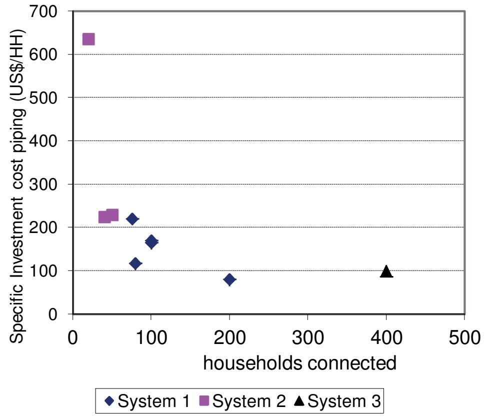


Figure 3.2 Specific piping investment cost (US$) per household using the system, arranged by system

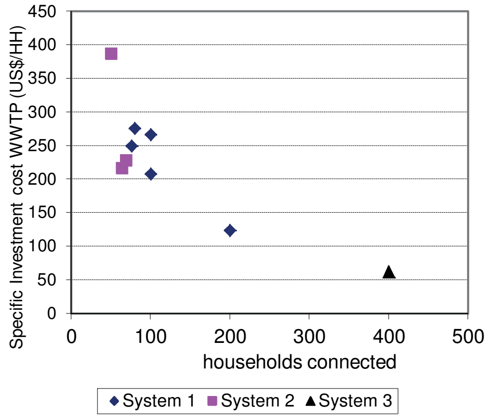


Figure 3.3 Specific WWTP investment cost (US$) per household using the system, arranged by system

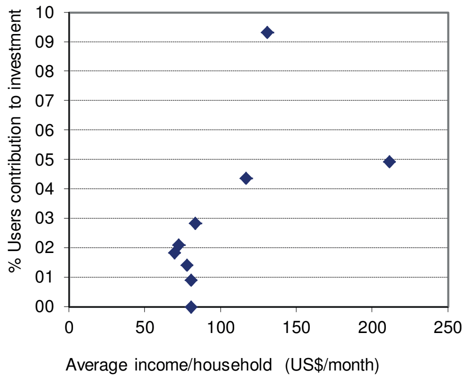


Figure 3.4 Percentage of contribution of users to the total investment costs

Presented findings are in line with previous studies which use a 1.5 times higher price for a separate black and grey water treatment system compared to a mixed wastewater sewer pipe (Kerstens et al., 2009). It is acknowledged that from this study it is not clear how costs will develop for bigger or smaller scales of System 3. However, decreasing specific costs with increasing household connections is plausible. The evaluated site of System 3 is designed for 400 households, but is currently operated only for 100 households. A follow-up visit in August 2011 to System 3 showed that more houses were getting connected. Therefore, for the investment costs for the sewer system the current sewer costs were multiplied by a correction factor of 3, following the calculated economy of scale in the evaluated systems via extrapolation.

3.3.2.2 Running costs (CAPEX + OPEX) and fees

In Table 3.3 the balance of costs (CAPEX + Operational Expenditures (OPEX)), benefits (from fees) and the cash deficit are presented for each site as well as the current fee/household/month and the fee that would be required to come to a full cost recovery including depreciation.

According the Sanimas (Sanitasi Masyarakat, or Sanitation by Community) approach (personal communication with BORDA) the level of fees should be based on all operational costs, being (a) daily operation and maintenance, small repairs, (b) local management and paid labour, (c) desludging, (d) large repairs and (e) replacement costs. Interviews with KSM, however, showed that only the first two (a and b) types of operational costs are seen as operational costs. As a result fees generally do not cover costs for desludging, large repairs or replacements. New connections are not paid from these fees, but are borne by the new household separately. The one exception where no fees are and will be collected is the case of System 3. The effluent of this system is expected to be of such good quality that it will be used as feeding water for a fish pond and that revenues from fish sale will pay for all types of costs of the service. As the fish pond was still under construction, this could not be verified and critical follow-up on the financial feasibility is recommended.

In addition it was found that fees were not adjusted during the years of operation, whereas inflation in Indonesia has been considerable, with an average value of nearly 7% since 2003 (http://dds.bps.go.id/eng/). It is recommended that in the determination of the fees all operational costs should be enumerated and yearly correction of fees following inflation is required.

Finally, it is found that operational costs (OPEX) are only a fraction (3-9%) of the total costs (CAPEX + OPEX). Table 3.3 therefore gives the (flat) fees that the users should be paying to meet all operational costs. Similar values of fees to meet full cost-recovery for a centralized system were confirmed in discussions with the director of Joint Secretariat (Kartamantul) of Yogyakarta. During that meeting it was confirmed that such values will not be accepted by users.

3.3.2.3 Financial and economical evaluation

Presented results show that at this point the capital costs are not borne by the users and could therefore be considered to be not financially viable from the point of view of a project manager. Table 3.4 shows that indeed only a small percentage of the investment costs are covered by the community. The major part of the capital cost is provided by central, provincial and kota (city) /kabupaten government, which is the same for the centralized sewerage services in the more affluent parts of the cities. Figure 3.4 shows, however, that there is a trend towards a higher contribution of the community to the investment costs from those with higher household incomes.

Table 3.3 Cash balances of the evaluated sites and actual and required fees for break-even

		CAPEX	OPEX	Fees	Cash Deficit	Actual fees	Required fee break-even incl. CAPEX
System	Project name	Million IDR/year				IDR/month/HH	
System 1	Minomartani	35.8	2.0	4.6	33.3	5,000	41,500
	Santan	29.0	1.2	4.8	25.4	5,000	31,500
	Kragilan	41.3	1.5	3.6	39.2	3,000	25,600
	Sukorejo	40.0	3.0	3.6	39.9	3,000	17,900
	Karangwetan	38.8	1.2	3.6	33.4	3,000	33,400
System 2	Gambiran	29.2	1.2	1.2	29.2	2,000	50,700
	Pajang	32.7	3.0	5.4	30.4	6000 per HHC [a] 500-1000 MCK	43,200
	Srenang	36.4	2.4	2.4	36.4	5000 per HHC 500-1000 MCK	50,600
System 3	Kepanjen Kidul	60.3	4.4	0.0	64.7	0	13,500

[a] HHC: household connection; other users pay 500 IDR/toilet visit and 1000 IDR/washing

Table 3.4 Origin of funds as a percentage for investment costs

Project	System	Central	Provincial	Kabupaten/ kota	BORDA	Users
Minomartani	System 1	27	0	53	15	4.9
Gambiran	System 2	33	0	66	0	1.8
Santan	System 1	30	0	60	0	9.3
Kragilan	System 1	26	0	64	7	2.8
Pajang	System 2	24	13	62	0	1.4
Srenang	System 2	22	13	63	0	2.1
suko arum	System 1	0	0	77	19	4.4
karang wetan	System 1	0	30	61	8	0.9
Kepanjen Kidul	System 3	0	91	9	0	0.0

In 2008 the World Bank calculated that the economic losses due to lack of access to sanitation of 43% of the Indonesian population were 6.5×10^9 US$/year, mostly as a result of accumulating health costs (53%) and (drinking) water costs (24%), whereas the yearly gains upon improving this situation would be 5×10^9 US$/year (WSP 2008). The Internal Rates of Return for the Indonesian Government to invest in providing sanitation to the not yet connected population using System 1, 2 and 3 are respectively 71%, 38% and 155%. Liang & van Dijk (2010) showed that the use of Decentralized Sanitation and Reuse (DESAR) systems in Beijing, China, is economically but not financially feasible as well. A similar study (Alam & Marinova, 2003) showed the often underestimated economic benefits for intervention in environmental protection. It is concluded that from a perspective of net economic gains, greater government investments in improved sanitation services would be a wise decision. Methods to also calculate all costs that must be covered to sustain an adequate service delivery are being developed and tested elsewhere (Fonseca et al., 2010).

It was found that fees are the same for all users and no progressive fees were applied (e.g. a higher fee for people with a higher income) or a fee based on a higher water consumption (to reflect the polluter pays principle). As shown in the next section, the poor, with an average income of less than US$2 per day, formed 28% of the household sample, while householders with a daily per capita income of US$7 or more formed 24%. Households in the first group tend to have only one tap and toilet, while households in the highest income category are likely to have more than two wastewater amenities and produce a much higher volume of wastewater than the first. Yet both categories contribute the same amounts to the investment and recurrent costs of the service. The currently adopted and promoted cost sharing system thus disproportionally benefits the better-off and it would be fairer to use a weighed system whereby households with 1-2 rooms or a “low-volume house” pay less than households with more rooms or with a medium- or high-volume house. Participatory rapid appraisal methods are an easy and well-accepted way to develop a more equitable and community-agreed tariff system (TTPS, 2009).

3.3.3 Evaluation by the user households

Figure 3.5 shows the results of the user households’ involvement in the different stages of the service and in the decisions on the type and level of the monthly fees per type of evaluated system. It is the intention to further segregate this data by socio-economic level and for women and men in a later study.

The involvement of the community at an early stage is highest for System 1, followed by System 2 and finally in the latest type of System 3 the community is far less involved in the planning and implementation phase, and only becomes more involved from the operational phase. In addition, for both System 1 and 2, the community determines the price for new connections, whereas for System 3 this is done by an external contractor. For all systems new connections are largely paid for by the users themselves.

The participatory implementation of DEWATS systems was found to enhance the process of acceptance and management of the applied technologies. Implementation of DEWATS systems in Indonesia is based on a demand-response approach, whereby only those communities showing willingness to participate in planning, training activities and to manage the costs and operation and management (O&M) are selected (Roma & Jeffrey, 2010). Because of the direct involvement of the community at – in principle – each keypoint of the project cycle (especially in local decision-making and in O&M, including financing management) these DEWATS systems are also known as Sanimas and the inclusion of local economic and social aspects is part of the widely accepted approach (TTPS, 2010). It seems that in the approach followed by BORDA and its local counterpart LPTP, who are responsible for the design and facilitation, a more intensive community involvement is applied compared to what was applied by the design party of System 3 (Konsultan Pyramida Utama).

Moreover, all heads and the majority members of the KSM were male, whereas women made more use of and are more involved in daily system operation. It is suggested to have more women involved in the KSM to include their experience as well. Further, improvements in informed decision-making to facilitate the recovery of costs and equity of financing are recommended.

The most important (perceived) benefit of the users of System 3 is improvement of the environment, whereas fewer benefits are seen in improved public health or improved water quality of dug wells. Possibly this relates to the fact that this community, in comparison with the other system users, makes only little use of dug wells and mostly relies on deep wells, which are less susceptible to contamination and have lower risks to health than dug wells. Alternatively, it could indicate a less effective awareness campaign delivered by the facilitator. In general, all visited communities were (very) satisfied with the DEWATS.

Figure 3.5 Results of the user surveys

3.4 Conclusions

Three different DEWATS used in Indonesia were reviewed, each with different technical designs, financial and economic parameters and different levels of community involvement. The summary of findings and the main conclusions on technical, financial-economic and community involvement are, respectively:

- The evaluated DEWATS systems all complied with the current Indonesian regulations, although these effluent standards are not very stringent compared to those of neighboring countries. System 3 (activated sludge) showed the best effluent quality on evaluated parameters COD, BOD, TSS, N and P whereas the other two systems were more or less equal to each other. High coliforms and nitrogen effluent concentrations pose a threat to public health and the environment without corrective measures. More attention to O&M and training and site specific constraints in the design phase are recommended. Although, system 3 (activated sludge) comes out most positive, further study of this type of system is recommended as only one system was evaluated and because it is identified as the most vulnerable system in the O&M phase;
- DEWATS servicing the smallest communities (System 1 Dig-Set-ABR-AF) showed the highest investment costs per household, followed by the system providing larger communities (System 2 Set-ABR-AF) and System 3 (Set- Act Sludge- Filter). This economy of scale effect is also seen in the piping system and the treatment section only and it is recommended to look for ways to increase the scale of DEWATS. The users usually only cover the direct operational costs. By not desludging they avoid having to pay for these costs, which they are expected to meet as well. Their share in capital costs, typically more than 90% of the overall costs of the service, is very limited and does not reflect differences in socio-economic levels. Government investment in DEWATS systems specifically and in environmental protection in general would have substantial economic benefits and rates of return. For sustained and equitable service delivery a greater share of the user communities in recurrent cost financing is needed. Better informed community decisions are needed on ways in which financing can reflect the variation in discharge more equitably ("polluter pays" principle). This also goes for the regular adjustment of fees to higher recurrent costs because of inflation and system aging;
- Systems 1 and 2 have a bigger involvement of the community in the planning and implementation phases of the projects compared to system 3. In all cases the communities acknowledge the positive effects of the system and show a great level of satisfaction with the system.

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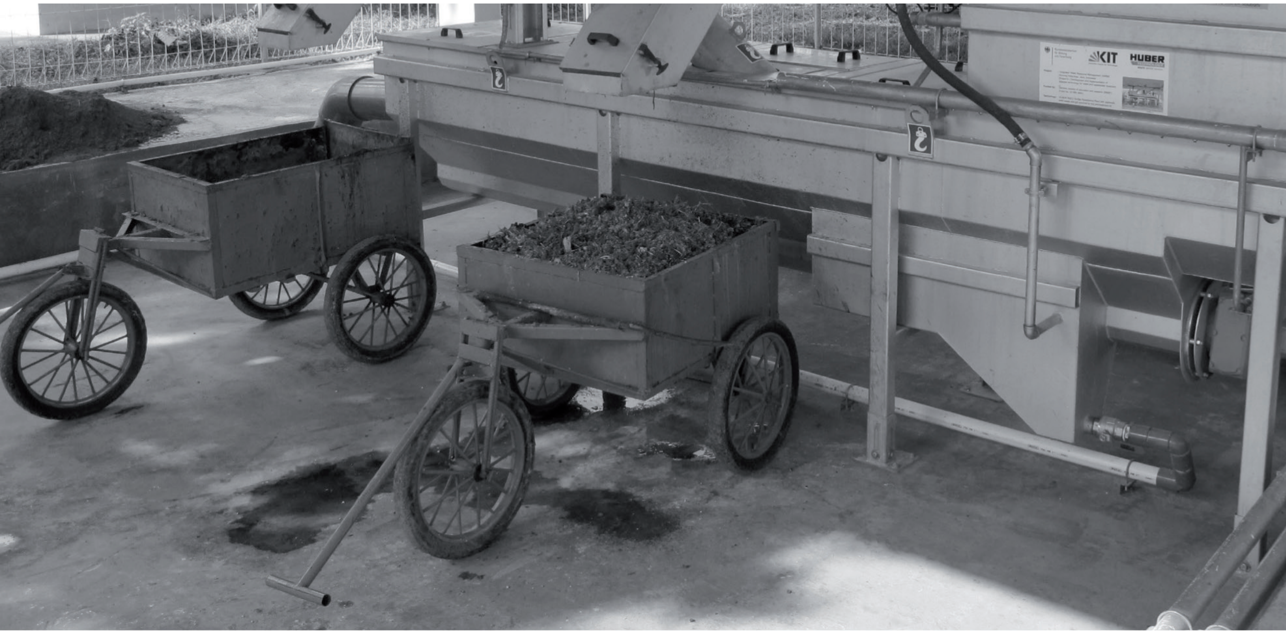
Chapter 4

Feasibility analysis of wastewater and solid waste systems for application in Indonesia



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Abstract

Indonesia is one of many developing countries with a backlog in achieving targets for the implementation of wastewater and solid waste collection, treatment and recovery systems. Therefore a technical and financial feasibility analysis of these systems was performed using Indonesia as an example. COD, BOD, nitrogen, phosphorus and pathogen removal efficiencies, energy requirements, sludge production, land use and resource recovery potential (phosphorus, energy, duckweed, compost, water) for on-site, community based and off-site wastewater systems were determined. Solid waste systems (conventional, centralized and decentralized resource recovery) were analyzed according to land requirement, compost and energy production and recovery of plastic and paper. In the financial analysis, investments, operational costs & benefits and total lifecycle costs (TLC) of all investigated options were compared. Technical performance and TLC were used to guide system selection for implementation in different residential settings. An analysis was undertaken to determine the effect of price variations of recoverable resources and land prices on TLC. A 10-fold increase in land prices for land intensive wastewater systems resulted in a 5 times higher TLC, whereas a 4-fold increase in the recovered resource selling price resulted in maximum 1.3 times lower TLC. For solid waste, these impacts were reversed – land price and resource selling price variations resulted in a maximum difference in TLC of 1.8 and 4 respectively. Technical and financial performance analysis can support decision makers in system selection and anticipate the impact of price variations on long-term operation. The technical analysis was based on published results of international research and the approach can be applied for other tropical, developing countries. All costs were converted to per capita unit costs and can be updated to assess other countries' estimated costs and benefits. Consequently, the approach can be used to guide wastewater and solid waste system planning in developing countries.

Keywords: wastewater, solid waste, financial analysis, technical analysis, developing countries, resource recovery

4.1 Introduction

The Millennium Development Goals (MDG) state that the proportion of people without access to sanitation facilities should be halved by 2015 compared to 1990. Nevertheless, a large fraction of the population in developing Asia currently lacks access to improved sanitation (ADB, 2012). In 2010, access to improved wastewater facilities in Indonesia was 56% (Ministry of Health, 2010) while progress reports suggest that Indonesia is not on track reaching the MDG's (WHO & UNICEF, 2014). National Health Surveys (Ministry of Health, 2013) show that less than 25% of households is served by a solid waste management system.

The vast majority of households in Indonesia with access to wastewater facilities rely on septic tanks (WSP, 2013a). A septic tank is the minimum treatment requirement in Indonesia (BPS, 2014) and minimum design standards for septic tanks are available (MoPW, 2000), yet rarely enforced. Consequently, 95% of septic tanks leak and result in the pollution of groundwater (WSP, 2013a). Community based systems or SANIMAS (Indonesian: *Sanitasi oleh Masyarakat*) comprising a community sanitation center or a simplified sewer system of small diameter pipes connected to an anaerobic baffled reactor, have been gaining grounds (Ulrich et al., 2009; Roma & Jeffrey, 2010; Reynaud et al., 2012a). By 2010, nearly 600 of such systems were implemented with 5,000 additional systems planned for the near future (Eales et al., 2013; Kearton et al., 2013). Evaluation of these systems (Reynaud et al., 2012b; Kerstens et al., 2012; Eales et al., 2013) confirmed the technical capabilities of the anaerobic systems to meet applicable effluent standards (MoE, 2003). However, challenges were identified such as the division of roles and responsibilities in technical and financial management, and the removal and safe disposal of sludge (Kerstens et al., 2012; Eales et al., 2013).

By 2012, only 12 centralized municipal wastewater treatment plants (WWTP) were in operation in Indonesia serving less than 1% of the population (USAID (United States Agency for International Development), 2006; Kearton et al., 2013). The systems utilized were (aerated) lagoons, UASB (Upflow Anaerobic Sludge Blanket), Rotating Bio Contactors (RBC's) and activated sludge systems (Kearton et al., 2013). Poor sewer network quality causes seepage of groundwater into the network, which dilutes the sewage and increases the flow to the treatment works (USAID, 2006). Connecting households to the sewer systems is a major problem (Whittington et al., 2000; Kearton et al., 2013) and requires institutional strengthening and advocacy (Winters et al., 2014). Several medium centralized WWT systems (serving 500 to 5,000 households), typically RBC's or Anaerobic Filters, were established in the past years (PDPAL-Banjarmasin, 2012) or are planned (Kearton et al., 2013).

Existing municipal solid waste (MSW) systems include the collection of waste from households by motorized or hand carts to a transfer station, followed by transportation to a landfill (TTPS, 2009; Aprilia et al., 2012). Between 2010 and 2014, 207 municipal landfills were constructed but only 132 have sufficient capacity until 2019 (MoPW, 2014a). The government is aiming for a 20% reduction of (urban) waste landfilled through the promotion of the "3R concept (Reduce, Reuse, Recycle)" (Bappenas, 2011), which has resulted in the construction of approximately 300 communal 3R stations by 2014 (MoPW, 2013a).

The lack of adequate wastewater systems combined with inadequate solid waste management is causing the contamination of both surface and ground waters (ADB, 2013) and thereby posing public health and environmental risks and economic losses (Hutton, 2013; Baum et al., 2013; Wright et al., 2013). Furthermore, the value of resources in wastewater and solid waste, such as energy, water, organics, nutrients and other recoverable products like plastic and paper, is being ignored. Resource recovery can benefit long-term operational and financial sustainability, while offering access to hygienic sanitation (Murray & Ray, 2010; Sasaki & Araki, 2013). Energy usage for conventional aerobic technologies contributes significantly to operational costs (Chernicharo, 2006) and in the absence of a stable power supply, sustainable service provision may be compromised (Lettinga, 2006). The predicted population growth and urbanization (BPS, 2013) will add pressure on space availability (related to population density), especially in urban areas (Aprilia et al., 2012). Consequently, the area footprint of facilities becomes an important parameter in system selection. In this paper, system selection covers collection, transport, treatment, disposal and resource recovery (Tilley et al., 2014).

This study provides a combined feasibility analysis of wastewater and solid waste technologies and combinations thereof. Both of these sanitation sub-sectors (wastewater and solid waste) aim to improve public health and the environment, and should therefore be addressed and solved simultaneously to achieve the desired quality of life (Ersoy et al., 2008; ADB, 2013). For that reason wastewater and solid waste management are often managed by one public authority e.g. a single ministry, as is the case in Indonesia and China (Yan et al., 2006; ADB, 2013). Moreover, both waste streams (water and solids) concern anthropogenic sources and are intrinsically related to human settlement development. Linking the feasibility of wastewater and solid waste technologies to (1) data on the population that has access to wastewater and solid waste facilities and (2) key residential features (urban/rural, land availability or population density) would therefore result in a sanitation decision support system and planning framework, showing the number of required systems and the associated costs. Data on access to sanitation, residential features and population development and prognoses for a wide variety of development countries are freely available (UNpopulation, 2012; Ministry of Health, 2013; BPS, 2014; DSM, 2014; NBSC (National Bureau of Statistics in China), 2014; WHO & UNICEF, 2014). However, despite the availability of general guidelines on system selection (TTPS, 2009) and a range of comparisons and evaluations on wastewater and solid waste systems (USAID, 2006; WSP, 2011; Aprilia et al., 2012; Eales et al., 2013; Kearton et al., 2013), a combined feasibility analysis of wastewater and solid waste systems under different residential conditions is lacking in scientific literature.

A second reason for an integrated wastewater and solid waste analysis is that the organic fraction of solid waste and wastewater can be treated using similar technologies such as digestion and composting (Zeeman & Kujawa-Roeleveld, 2011). In addition, energy consuming wastewater processes (e.g. aerobic technologies) may be combined with energy producing (anaerobic) solid waste or wastewater treatment processes resulting in net energy producing systems (Kujawa-Roeleveld & Zeeman, 2006; Ersoy et al., 2008; Kerstens, De Mes, et al., 2009; Meinzinger et al., 2009). Insights into potential synergy for the treatment of wastewater and solid waste flows may thus result in more favorable financial feasibility and consequently accelerate sanitation development.

This paper aims to provide an analysis of selected wastewater treatment and municipal solid waste systems, including the financial and environmental performance of these systems under Indonesian conditions. It is hypothesized that a wastewater and solid waste system selection can be based on a small number of readily available parameters (residential density, urban/rural features). By including both investment and operational costs and benefits, the proposed system selection framework can be combined directly with life cycle costs, thereby allowing for the development of a principle framework for wastewater and solid waste planning and costing in developing countries.

The analysis was based on international literature and includes a comparison of different technological systems according to: (1) removal efficiency of COD, BOD, nitrogen, phosphorus and pathogens, (2) sludge production, (3) energy consumption, (4) area requirement and (5) resource recovery potential (phosphorus, energy, duckweed, compost and water). Secondly a financial analysis was performed focusing on a comparison of investment and operational costs as well as the potential benefits accrued from resource recovery. Subsequently the total lifecycle costs (TLC), comprising investment and operational costs minus potential benefits over a 20 years operation time, were evaluated.

4.2 Material and Methods

Wastewater systems are first compared based on technical performance and the potential to comply with applicable regulations in order to minimize public health and environment risks. Following Indonesian Ministry of Public Works guideline definitions, three types of systems are evaluated – on-site systems (typically serving 1-5 households), community based systems (CBS; typically serving 50-150 households) and off-site systems (typically serving more than 500 households). Off-site systems are further split into medium centralized systems (< 5,000 households) and centralized systems (> 10,000 households). For off-site systems 10 technologies that are currently applied or under consideration for application in Indonesia were evaluated. For on-site and CBS the currently applied septic tank and anaerobic baffled reactors were included in the analysis (see Table 4.1 and Table 4.2).

Four types of municipal solid waste systems currently applied in Indonesia (MoPW, 2013b) were compared and evaluated according to space requirement and potential for resource recovery (see Table 4.3). In the subsequent financial analysis, the costs and benefits of WWT and MSW systems and technologies are compared. The applicability of the evaluated technologies was compared for varying land conditions and varying recovered resource selling prices.

4.2.1 Methodology for the determination of WWT technical performance

4.2.1.1 Wastewater composition and quantities

The same wastewater composition was assumed for all systems and technologies. The technical evaluation is performed based on COD (Chemical Oxygen Demand), N (nitrogen) and P (phosphorus) generation rates per capita per day as reported by Almy (2008) and a COD/BOD (Biological Oxygen Demand) ratio based on Meinzinger & Oldenburg (2009). Water consumption is based on the MoPW guideline (MoPW, 2011), applying an 80% wastewater return factor (DKI, 2005). Household size was defined as 5 persons (Almy, 2008). To determine black water concentrations, a daily toilet usage frequency of 6 per person (Kujawa-Roeleveld, 2005) with a flush volume of 6 l/p/d (De Graaff et al., 2011) was used. Applied COD values correspond with those reported elsewhere (Meinzinger & Oldenburg, 2009), corrected for COD of toilet paper (De Mes, 2007), which is typically not used in Indonesia (Reynaud et al., 2012a). Calculated wastewater composition and quantities are shown in Table 4.4 and further detailed in Table A4.1 of the Appendix Chapter 4.

4.2.1.2 Investigated WWT technologies

Selection of investigated technologies was first of all based on compliance with the definition of the Indonesian statistical bureau (BPS) which describes a latrine, a goose neck (*leher angsa*) and a septic tank or sewer connection with treatment (BPS, 2014). The second criterion was the already applied or planned status in Indonesia, based on the study of centralized WWTP by USAID (2006) and developments in the Jakarta Master Plan (JICA, 2012). The third criterion was the appropriate level of technologies and collection systems in view of the general level of technology in Indonesia. Poor operation and maintenance of facilities is a recurring bottleneck (USAID, 2006; Kearton et al., 2013). Thus, “new sanitation concepts” separating grey water, black water or feces and urine for enhanced resource recovery and applying vacuum transport systems (Zeeman & Kujawa-Roeleveld, 2011) are excluded because these approaches have not been proven on a large scale in developing countries. Separation of black water is only applied on a household level using septic tanks, which is already common in Indonesia (WSP, 2013a). Finally resource recovery technologies are selected that (1) could be added to technologies in the wastewater sludge processing line (phosphorus, energy and compost) or (2) recover multiple resources in the water line (energy, duckweed, compost). In Table 4.1 the selected WWT systems are described by (I) typical scales (households), (II) features of sewer systems, (III) sludge management, (IV) selected technologies excluding and (V) including resource recovery.

Table 4.1 Scale, sewer systems, sludge management, technologies and potential recovered resource products of selected WWT systems

Systems	On-site	Community Based (CBS)	Off-site systems	
			Medium centralized	Centralized
I. Scale: number of households [a]	1	50-150	500-5,000	> 10,000
II. Sewer system	None	simplified sewer system		(pumped) sanitary system
III. Sludge management	Centralized sludge processing facility (IPLT [b])		At the location of WWTP	
IV. Technologies excluding resource recovery	Septic Tank (treating black water)	Anaerobic Baffled Reactor + Filter (ABR + AF)	1. Anaerobic filter (An. Fil) 2. Aerated lagoon 3. Conventional Activated Sludge (CAS) 4. CAS + enhanced (N) and (P) removal (CAS + N&P) 5. Aerobic Granular Sludge (AGS) 6. Membrane Bioreactor (MBR)	
V. technologies including resource recovery	Composting of IPLT [b] sludge		7. Technology 4-6 with additional: o Sludge digestion to energy; o Struvite Crystallization from centrate o Dewatered Sludge composting 8. UASB-DW-RBC [c]; producing energy, duckweed (proteins), compost	

[a] based on MoPW (2013c), Kerstens et al. (2012), Reynaud et al. (2012b), and MoPW (2014a)

[b] IPLT*: Instalasi Pengolahan Limbah Tinja;* a sludge processing facility

[c] UASB (Upflow Anaerobic Sludge Bed), DW (Duckweed Ponds); RBC (Rotating Bio Contactor)

Both CBS and medium centralized systems apply simplified sewer systems with lower costs compared to a conventional or pumped sanitary system because of reduced pipe length, smaller diameters, shallow installation and simple inspection pits (Mara & Broome, 2008; Van Buuren, 2010). Sanitary systems transport wastewater separately from rainwater and are particularly applicable in tropical regions with high rainwater run-off. These systems may require additional pumping depending on slope and distance to the final treatment plant (Van Buuren, 2010).

Sludge management for on-site and community based systems in Indonesia is frequently performed at sludge processing facilities (IPLT) where collected sludge is delivered by vehicles (MoPW, 2013c; WSP, 2013b). The Imhoff Tank is typically applied as a first step of the IPLT followed by a simple aeration step (MoPW, 2013c).

Applied removal efficiencies for on-site, CBS and the off-site technologies are presented in Table 4.2. Details of design parameters for each technology and examples of mass balances are described in Section 2 of the Appendix Chapter 4. The conclusions on removal efficiencies assume that systems are designed, constructed, operated and maintained correctly. Current practice in Indonesia, however, shows that operation and maintenance of wastewater treatment systems is often poorly managed (ADB, 2013; Kearton et al., 2013)

The impact of variations in removal efficiencies was therefore analyzed by including an efficiency removal reduction on values in Table 4.2 of (1) 10% for COD and BOD, (2) 5% for nutrients

(N&P), and (3) 1 log for coliforms. The 10% COD and BOD removal efficiencies were based on the spread of reported removal efficiencies in a UASB system by Chernicharo (2006). The 5% reduced nutrient removal efficiency variation was based on the expected total N effluent variation in biological nutrient removal systems (EPA, 1993). The reduced log removal was based on Tchobanoglous et al. (2003).

Table 4.2 Applied removal efficiencies for selected wastewater treatment technologies [a]

Technology	COD [b]	BOD [b]	Total Nitrogen (TN) [b]	Total Phosphorus (TP) [b]	Coliforms [b]
Septic Tank [c]	60%	70%	15%	5%	~ 1 log
ABR + AF	80%	85%	15%	5%	~ 2 log
Anaerobic Filter	80%	85%	15%	5%	~ 2 log
CAS	88%	96%	73%	29%	~ 4 log
Aerated lagoon	88%	96%	73%	29%	~ 4 log
CAS + N&P	89%	97%	90%	67%	~ 4 log
AGS	89%	97%	90%	67%	~ 4 log
MBR	91%	98%	91%	67%	100%
UASB-DW-RBC	93%	95%	97%	73%	~ 4 log

[a] In case additional resource recovery technologies are added to the CAS + N&P, AGS and MBR, the same removal efficiencies are obtained as in cases without resource recovery

[b] Applied removal efficiencies are based on literature, described in section 2 of the Appendix Chapter 4

[c] Septic tank removal efficiencies relate to black water treatment only

4.2.2 Municipal Solid Waste (MSW) Management Systems

4.2.2.1 Waste generation and composition

Solid waste mass balances were derived based on a specific waste production of 2.5 l/p/d for rural and 2.75 l/p/d for urban areas with a density of 0.25 ton/m^3 (BSN, 1995), resulting in a daily MSW production rate of 0.63 and 0.69 kg per person for rural and urban areas, respectively. These rates correspond with those reported elsewhere (Bhattacharya et al., 2005). Applied waste density at the transfer station to determine the number of vehicles necessary for waste transport to a landfill was 0.34 ton/m^3 (KNLH, 2008). The applied Organic Solid Waste Fraction (OSWF) was 59% (mass) (KNLH, 2008). Plastic and paper content were obtained from Aprilia et al. (2013).

4.2.2.2 Investigated MSW systems and determination of process features

Four MSW systems were defined with each system comprising a full chain of activities: collection, transfer and transport of waste, treatment and disposal (Table 4.3). System I is a conventional system, in which all collected waste is landfilled and no recovery is applied. System II applies waste (organic, plastic and paper) separation on a centralized level (location of landfill). Plastics and papers are packed and recycled (Sasaki & Araki, 2013) and the OSWF is treated using composting only. System III is similar to the second system, but applies digestion prior to composting of OSWF. System IV applies OSWF composting and plastic and paper recovery at a decentralized or community level. For rural areas only decentralized 3R systems were assumed, as direct reuse of produced compost is envisaged. Appendix Chapter 4, Section 4 provides details on design parameters, mass balances and a full cost breakdown for the analyzed systems.

Table 4.3 Applied characteristics and efficiencies of selected Municipal Solid Waste (MSW) systems

System / Activity	Conventional	Applying 3R (Reduce Reuse and Recycle)		
		Centralized 3R		Decentralized 3R
		Composting	Dig & Comp	
Collection from households	Motorized vehicles: 3 trips/day each [a]			
transfer station	Covered station with 2 (+1) containers [a].			3R transfer station [a]
Transport to centralized facility	Armroll truck [a] (3 trips/day each)			
Fraction of Plastic and Paper recovered	0%	75% [b]; remaining part is landfilled.		
Organic solid waste fraction (OSWF) separation efficiency	0%	75% [c] centralized		75% [c] decentralized
Fraction of separated OSWF digested	0%	0%	63%[d]	0%
Compost production coefficient; using open windrowe	0%	0.35 kg compost/kg organic waste[f]		
Disposal in landfill	15 m depth, 20 years lifetime, with 4 stages, covering 5 years. In the first phase land for the full period and facilities (office, weighing bridge, vehicles, leachate treatment plant) [g] are arranged			

[a] MoPW (2013b), TTPS (2009); [b] KLH (2012); [c] Aprilia et al. (2012); [d] calculated, shown in the Appendix Chapter 4, Section 4; [e] Compost is piled in rows and work without forced aeration and waste gas collecting (Saveyn & Eder, 2014); [f] MoPW (2013b) and calculated as shown in the online supplementary information, Section 4; [g] MoPW (2013b), TTPS (2009)

4.2.3 Considerations for the financial analysis

Costs of sewer systems serving CBS are based on Kerstens et al. (2012). The methodology to determine the costs for off-site sewer systems is further explained in Section 3 of the Appendix Chapter 4. For “greenfield” situations a 50% reduction in sewer construction costs was applied because sewer construction can be combined with other development (Rioned, 2007). Investment costs for on-site and CBS sludge management facilities are based on MoPW (2013c) and Kerstens et al. (2012). AGS, MBR and CAS investments are based on prepared designs by Royal HaskoningDHV and adjusted to local conditions in order to correct for price level differences (Moore & Mathew, 2012) (see Appendix Chapter 4, Section 5). Costs for UASB, RBC, AF and aerated lagoon are based on Van Buuren (2010) and Chernicharo (2006). All treatment prices are estimates and may vary by up to 30%. Price variations are the result of a combination of factors. Site labour costs remains one of the key drivers, while the sourcing of materials, including the use of imported mechanical and electrical engineering plant, can have a profound effect on prices. In addition, procurement and contractual arrangements can also have a substantial effect on costs (Moore & Mathew, 2012).

Collection and landfill costs were based on MoPW guidelines (TTPS, 2009; MoPW, 2013d, 2014b). MSW 3R costs were based on MoPW (2013b) and are further explained in Section 4 of the Appendix Chapter 4. Costs include studies/design, advocacy, campaigns, institutional training, construction and supervision. Labour (technical and institutional) costs are based on function group (MoPW, 2013b) and include both technical and institutional costs. Operational costs include replacement costs of collection vehicles (TTPS, 2009), whereas all other facilities (both WWT and MSW) are assumed to have an economic lifetime of at least 20 years. An overview of applied prices in the financial analysis is shown in Table A4.4 in Section 5 of the Appendix Chapter 4.

During this study, first hand information on prices in Indonesia was obtained during site visits and through discussions with involved stakeholders (see Section 5 of the Appendix Chapter 4).

Because reported land and product selling prices varied considerably, the impact of these price variations on the total lifecycle costs was further evaluated.

The applied currency conversion rates (July 2013) were 10,000 Indonesian Rupiah (Rp)/US$ and 13,200 Rp/Euro (XE-currency, 2014). TLC are the total costs in a defined lifetime and include Capital Expenditures (CAPEX) at the start (t=0) or at a later year (t=l) and net Operational Expenditures (OPEX). Net OPEX are calculated by subtracting benefits from operational costs. The TLC was determined using the Net Present Value (NPV), in which the total costs are discounted back to its present value. The use of NPV to determine total costs was applied by Starkl et al. (2010) and Hauger et al. (2002). A 5% inflation (I), based on the 2009-2013 average rate (Inflation.EU, 2014) and a 6% discount rate (D) (Global-rates.com, 2014) for 20 (n) years are applied in Formula (1):

$$TLC = \mathrm{CAPEX}_{t=0} + \mathrm{CAPEX}_{t=l} \times (\frac{1+I}{1+D})^{l} + \mathrm{Net\ OPEX} \times \frac{(\frac{1+I}{1+D})^{n} - 1}{(\frac{1+I}{1+D}) - 1} \qquad \text{(Formula 1)}$$

4.3 Results and Discussion

4.3.1 Wastewater treatment systems

4.3.1.1 Effluent values of systems and technologies

Table 4.4 shows the calculated influent and effluent composition and three effluent standards for comparison: the minimum (MoE, 2003), the more stringent Jakarta (DKI) standard (DKI, 2005) and the Indonesian irrigation standard (MoPW, 2001).

No effluent requirements apply for septic tank effluents, whereas the ABR + AF (CBS) should comply with the 2003 regulation (MoE, 2003). Both the ABR+AF and the AF effluent comply with this minimum standard for discharge, as also reported by Kerstens et al. (2012) and Reynaud et al. (2012b). Cities may determine their own (more stringent) standards for off-site systems. The standard for Jakarta (DKI) is shown in Table 4.4, as an example. The Anaerobic Filter does not comply with the DKI standard, whereas all other off-site technologies do. The CAS N&P, AGS and MBR systems do comply with the irrigation standard, whereas the UASB-DW-RBC effluent exceeds the recommended BOD value. The MBR complies with the Coliform standard, whereas for the remaining systems post treatment should be considered for safe use of effluent for irrigation, applying chlorination, ozonization, UV radiation or filtration (Chernicharo, 2006).

Lower removal efficiencies result in higher effluent values (see values in brackets in Table 4.4). Table 4.4 shows that the conclusion on meeting the minimum standards (MoE, 2003) does not change since effluents of all technologies, except the septic tank, do comply with the minimum standards. Except for the UASB-DW-RBC system, the DKI standard will not be met for any system in terms of COD with the assumed lower removal efficiency. The impact of assumed reduced removal efficiencies will also impact the conclusions on meeting the irrigation standard, as none of the technologies complies with COD, BOD and P standards. The assumed reduced Coli removal confirms the need for effluent disinfection for all systems (except for the MBR) to assure a hygienic practice.

Table 4.4 Calculated influent and effluent values for selected technologies in comparison with three different effluent standards; values in brackets for COD, BOD, N and P show the result of reduced removal efficiencies of 10% (COD, BOD), 5% (N&P) and 1 log (Coli)

Parameter (unit)	Influent	Effluent									Effluent standards		
		Septic Tank	ABR+AF	An. Filter	UASB-DW-RBC	CAS	Aerated lagoon	CAS N&P.	AGS	MBR	Minimum [a]	DKI [b]	Irrigation [c]
COD (mg/l)	604	363 (387)	121 (169)	121 (169)	41 (98)	70 (123)	70 (123)	66 (120)	66 (120)	52 (107)	-	80	100
BOD (mg/l)	302	127 (144)	45 (71)	45 (71)	15 (44)	11 (40)	11 (40)	10 (39)	10 (39)	7 (37)	100	50	12
NKj (mg/l)	91	77 (78)	77 (78)	77 (78)	1 (6)	4 (8)	4 (8)	4 (8)	4 (8)	3 (7)	-	10	20 (as TN)
NO_3-N (mg/l)	0	0 (0)	0 (0)	0 (0)	2 (2)	21 (22)	21 (22)	5 (6)	5 (6)	5 (5)	-	-	-
TP (mg/l)	15	14 (14)	14 (14)	14 (14)	4 (4)	11 (11)	11 (11)	5 (6)	5 (6)	5 (6)	-	-	5
Coli (U/100ml)	10^8	10^7 (10^8)	10^6 (10^7)	10^6 (10^7)	10^5 (10^6)	10^5 (10^6)	10^4 (10^5)	10^4 (10^5)	10^4 (10^5)	0 (10^1)	-	-	$1x10^4$

[a] Minimum standard (MoE, 2003); [b] (more stringent) Jakarta standard (DKI, 2005); [c] irrigation standard (MoPW, 2001)

COD, BOD, nutrients and pathogen effluent values for septic tanks are considerably higher than all other technologies. A relation between the use of (poor performing) septic tanks and eutrophication and high pathogen levels of water bodies near cities are eminent in Indonesia (Hart et al., 2002; Fach & Fuchs, 2010; Fulazzaky, 2010; Suharyanto & Matsushita, 2011). Septic tank performance can be improved using the UASB-septic tank (Lettinga et al., 1991). Alternatively, post treatment removal of remaining COD, N and P can be applied in a (medium) centralized system. Due to solids removal, chances are decreased for clogging and simpler and less expensive small bore sewer systems can be applied (Mara & Broome 2008;Van Buuren 2010). However, septic tank effluent post -treatment may be hindered in the Indonesian context:

- Because most septic tanks are not sealed and septic tanks require sealing (WSP, 2013a), which is a private (and costly) investment in Indonesia (WSP, 2011);
- Besides the management of a small bore sewer system and post-treatment plant, septic sludge management is needed;
- Minimum COD/N (EPA, 1993) and COD/P ratios (Janssen et al., 2002) for denitrification and biological P-removal apply if a CAS is used as post treatment. COD removal in a septic tank can jeopardize this ratio. COD is not a limiting factor in the removal of N and P using duckweed ponds + RBC as post treatment.

Table 4.4 shows high effluent ammonium values for CBS systems, which is in line with reported values (Reynaud et al., 2012a; Kerstens et al., 2012). Calculated CBS effluent P values (Table 4.4) are similar to measured values by Reynaud et al. (2012a) (15 mg/l PO_4-P), but higher than measured values by Kerstens et al. (2012) (4 mg/l PO_4-P). Reynaud et al. (2012a) attributed high P values to low water consumption. The lowest effluent values for N and P in this study were calculated for the UASB-DW-RBC technology (Table 4.4).

4.3.1.2 Energy consumption and production of technologies

A comparison of energy consumption and production of the described off-site technologies is presented in Figure 4.1. Septic tanks and ABR+AF require no energy input. Energy consumption for the IPLT treatment was calculated as 0.8 kWh/cap/year in this study. In the anaerobic filter organic matter is converted to biogas. A yearly per capita biogas production of 7 m^3 is calculated applying the same approach as for the UASB mass balance (see Section 2 of the Appendix Chapter 4). Since 1 m^3 of biogas equals 6 hours of 60-100 watt bulb light (Almy, 2008), collection of the biogas produced by 100 people would be sufficient to have lighting for 12 h per day. However, biogas from anaerobic filters is typically not used or flared (Kerstens et al., 2012) and thus contributes to greenhouse gas emissions and corresponding negative environmental impacts (Aye & Widjaya, 2006). The use of the produced biogas could result in a positive energy balance.

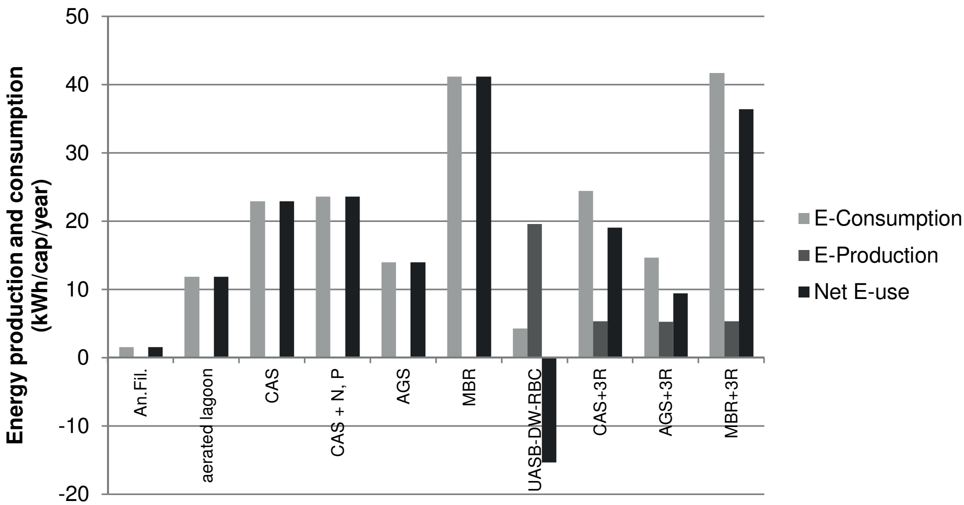

Figure 4.1 Calculated per capita values for energy (E) consumption, production and Net E-use (Consumption-Production) for off-site systems (shown as negative values if it concerns a net energy producing system)

Only the UASB-DW-RBC technology system produces net energy. The calculated fraction of incoming COD converted to methane is 0.47 (Section 2 of the Appendix Chapter 4) and is approximately 1.5 times the value of 0.31 and 0.33 reported by, respectively, Draaijer et al. (1992) and Kooijmans et al. (1986). The lower measured values in comparison to the calculated value could be related to methane leaving the reactor through the effluent as described by Kooijmans et al. (1986). Furthermore, produced biogas may not be captured properly due to leaking gas caps (USAID, 2006). Therefore, presented energy production may be optimistic and a lower value (~33%) could be expected in practice. In that case, however, the system still produces 9 kWh/cap/y.

Figure 4.1 and Table A4.5 of the Appendix Chapter 4) shows 40% and 67% energy consumption reduction for the Aerobic Granular Sludge compared to, respectively, the Conventional Activated Sludge and MBR, which is in line with full scale cases (De Bruin et al., 2013; Pronk et al., 2015). Sludge digestion for the Conventional Activated Sludge +3R, Aerobic Granular Sludge + 3R and MBR+3R technologies, results in a recovery of 22%, 36% and 13% of the consumed energy, respectively (see Table A4.5 of the Appendix Chapter 4).

4.3.1.3 Sludge and compost production

Table 4.5 shows the calculated sludge and compost production values for the selected technologies. Agricultural use of digested sludge or fecal sludge has long been practiced since the sludge may act as source of essential crop nutrients, stimulates microbial activity, immobilizes toxic elements in soil, improves soil structure, minimizes soil erosion (Tesfamariam et al., 2013) and contains essential elements for crop production (C, N, P, K and Ca) (Dobermann & Fairhurst, 2000). At the same time (1) health, (2) social acceptability and (3) environmental complications are associated with the application of digested sludge (Koné et al., 2007; Starkl et al., 2010; Tesfamariam et al., 2013), which is further elaborated in Section 7 of the Appendix Chapter 4.

4.3.1.4 Recovery of proteins, N, and P via duckweed

A duckweed production of 15.5 kg TS (Total Solids) per capita per year is calculated for the UASB-DW-RBC technology system. Duckweed is an excellent protein rich (20%) feedstock for aquaculture in Indonesia (Journey et al., 1993; Abery et al., 2005). Furthermore, duckweed has reported N and P contents of, respectively 3% and 0.5% of the TS (Korner & Vermaat, 1998; El-Shafai et al., 2007; Bal Krishna & Polprasert, 2008) and could be used as organic fertilizer (Journey et al., 1993). In addition, Duckweed ponds can considerably reduce odor emission (Kerstens et al. 2009). In the current study no post-treatment of the CBS (ABR+AF) effluent is applied, because the effluent already complies with the minimum standard, but application of duckweed ponds can be considered if the revenues from duckweed are higher than associated additional land costs.

P-recovery: In the Aerobic Granular Sludge, MBR and Conventional Activated Sludge + 3R approximately 0.35 kg P/cap/year (Table 4.5) is recovered via struvite precipitation and sludge reuse, which corresponds with 48% of the yearly P produced per capita. De Graaff et al. (2011) determined a P-recovery of 0.22 kg P/cap/year through effluent struvite precipitation and 0.16 kg P/cap/year sludge production for a UASB treating concentrated black water. The yearly P-recovery from duckweed and attached biomass was calculated as 0.31 kg P/cap/y, whereas the amount of P recovered from composted UASB and RBC sludge was calculated as 0.04kg P/cap/y.

Land use: The land use in the UASB-DW-RBC technology is about 400 times higher than the smallest footprint (AGS or MBR system) (Table 4.5), whereas the footprint of aerated lagoons is a tenfold that of the AGS and MBR. The effect of land use on lifecycle costs is further analyzed.

Table 4.5 Calculated [a] process production values and land use for selected WWT technologies and Sludge treatment (IPLT) for septic- and ABR+AF-sludge

Parameter	Unit	Septic	ABR+AF	IPLT [b]	An.Fil.	Aerated lagoon	CAS	CAS + N, P	AGS	MBR	UASB-DW-RBC	CAS +3R	AGS +3R	MBR +3R
Sludge prod.	l/cap/y	35.0	35.0	15.3	19.3	30.0	53.4	59.0	53.7	58.7	-	-	-	-
Sludge TS prod.	kg/cap/y	3.9	3.9	3.1	3.9	6.0	10.7	11.8	10.7	11.7	-	-	-	-
Compost prod.	kg/cap/y			4.4	-	-	-	-	-	-	3.6	16.9	14.7	16.8
Struvite prod	kg/cap/y	-	-	-	-	-	-	-	-	-	-	0.99	0.82	0.95
P struvite	kg/cap/y	-	-	-	-	-	-	-	-	-	-	0.22	0.19	0.21
P compost	kg/cap/y	-	-	0.0	-	-	-	-	-	-	0.01	0.09	0.12	0.14
Total P-prod	kg/cap/y	-	-	0.0	-	-	-	-	-	-	0.01	0.32	0.31	0.36
Land use	m^2/cap	0.0	0.20	0.00	0.09	0.29	0.04	0.05	0.03	0.03	4.70	0.06	0.03	0.03
Water prod.	m^3/cap/y	-	-	-	-	-	-	-	-	50	-	-	-	50
Duckweed	kg/cap/y	-	-	-	-	-	-	-	-	-	15.5	-	-	-

[a] Calculated based on approach described in the Appendix Chapter 4, Section 2

[b] IPLT: *Instalasi Pengolahan Limbah Tinja:* sludge processing facility

Water production: Only in the MBR is water (50 m³/cap/y; Table 4.5) produced that is directly safe for irrigation, greening (Kerstens et al. 2009) or as a source for drinking water production (Cikarang Estate, 2014). The feasibility of reusing this water depends, among others, on the distance to agricultural land or reservoirs and user acceptance. An evaluation of these aspects is beyond the scope of this study.

4.3.1.5 Financial analysis of WWT

Figure 4.2 shows the per capita TLC after 20 years of operation, subdivided by investments (CAPEX sewer, treatment and land) and Net operational costs. The numeric values of each system and technology are presented in Table A4.6 and A4.7 of the Appendix Chapter 4.

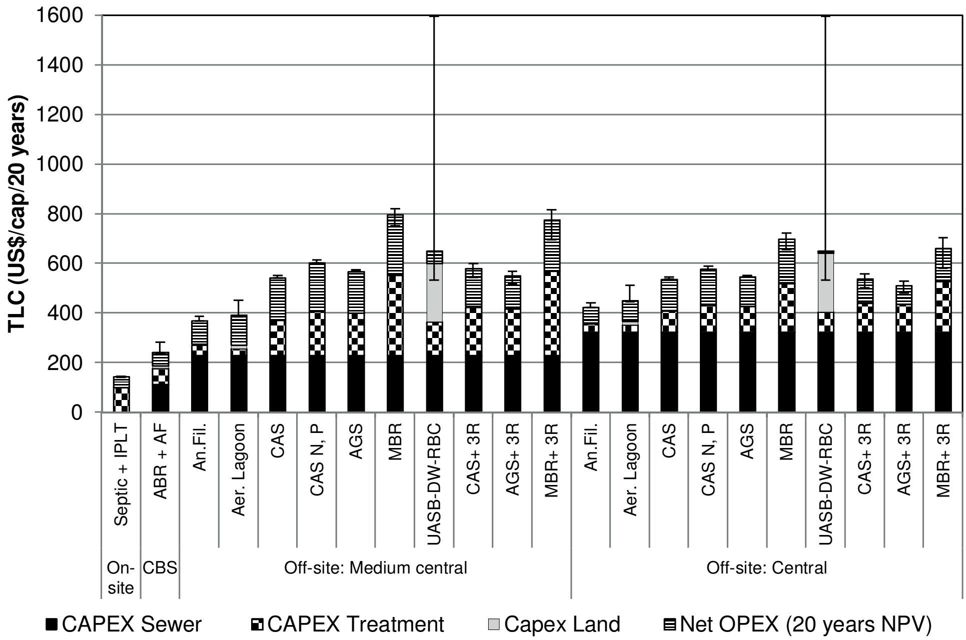

Figure 4.2 Per capita Total Lifecycle Costs (TLC) comprising sewer, treatment and land investments (CAPEX) and 20 years of NET OPEX (as NPV) per system and technology. Error bars show maximum and minimum values of variations in land prices (factors 0.5 to 5) and recovered resource selling price (factors 0.5 to 2)

Figure 4.3 shows the operational costs and benefits per parameter of on-site, CBS and medium centralized systems. The figure illustrates that costs for labour amount on average to 36% of the operational costs with the lowest percentage (22%) for the simpler systems (anaerobic filter) and the highest percentage (53%) for the more complicated systems (MBR). Energy costs for aerobic systems (excluding Anaerobic Filter and UASB-DW-RBC) contribute to about a quarter of the operational costs (range 17% to 28%). An average of 17% of the operational costs is required for maintenance of the selected off-site WWTPs (range 4% to 22%), with the lowest contribution for the anaerobic filters and aerated lagoon. The sewer maintenance costs contribute on average 16% of the costs (range 9% to 27%), with the anaerobic filters and aerated lagoon showing the highest contribution.

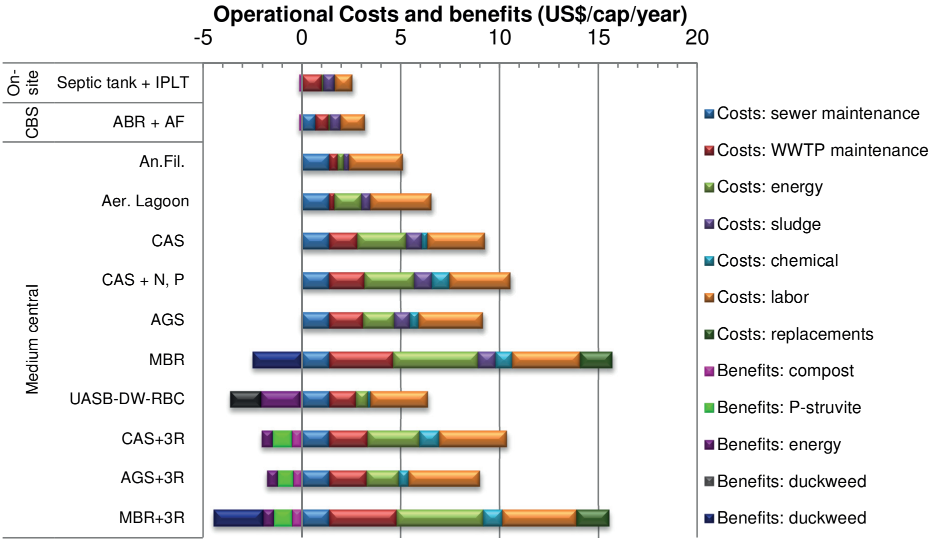

Figure 4.3 Per capita yearly WWT operational costs (positive values) & benefits (negative values) per system and technology

On-site and CBS systems have the lowest investment, Net OPEX and TLC (Figure 4.2 and Figure 4.3 and Table A4.6 and A4.7 of the Appendix Chapter 4). The indexed TLC of on-site (used as reference; set as 100), CBS, off-site (in existing areas) and off-site (green field) are, respectively, 100, 169, 398, 302. The average off-site costs are about 2.4 and 3.9 times higher than, respectively CBS and on-site technologies. However, off-site systems have higher removal efficiencies and better effluent quality (Table 4.4). Investments and operational costs of the anaerobic filter and aerated lagoon are considerably lower than those of other off-site systems even considering additional costs of a flaring system for an anaerobic filter (1-2% additional investment). The MBR, which is proposed in Jakarta (JICA, 2012), shows the highest investment, operational costs and TLC.

The TLC of off-site systems in green field situations is 75% of that in existing areas due to 50% lower sewer costs (Rioned, 2007) and therefore off-site systems become more financially attractive in new urban developments.

The percentage of sewer system costs on the total costs for medium centralized and centralized systems are respectively 54% and 71%. The lower sewer costs for medium centralized systems are the result of (1) smaller applied diameters and pipes, (2) lower traffic impediments (Mara & Broome, 2008; Van Buuren, 2010), but also (3) the result of the economy of scale effect of medium centralized treatment systems, as these have higher per capita costs (Starkl et al., 2012). The ratio between the off-site system with the lowest TLC (Anaerobic filter) and the highest TLC (MBR), for medium centralized and centralized system is, respectively, 2.15 and 1.65 (Table A4.6 and A4.7 in the Appendix Chapter 4). The smaller ratio for the centralized system compared to the medium centralized system suggests that for a centralized system additional treatment performance can be achieved at relative lower costs. The effect of material type and pipe bedding was not analyzed, however it is known that optimization of these construction aspects can lower sewer system construction costs (Petit-Boix et al., 2014).

Land costs (LC) and anticipated revenues from the sale of recovered products (Prod) can vary considerably and thus impact the TLC. Figure 4.2 showed the maximum and minimum values of each system and technology by changing either land price or recovered resource selling price. Figure A4.9 in the Appendix Chapter 4 shows the impact of each modelled land cost and product selling price on the TLC (excluding sewer investment and operational costs).

The effect of varying land prices is highest for the land intensive technologies (UASB-DW-RBC and aerated lagoons) with nearly a factor 5 difference between the highest and lowest analyzed land price for the UASB-DW-RBC. The reference land price was 50 US$/m^2, whereas several locations in Jakarta have land prices of 2000 US$/m^2 (KNI, 2014), which favors compact technologies. Development of a technology for the application of autotrophic nitrogen removal in the water line (Hendrickx et al., 2012) could favour the application of UASB technology, also in areas with high land prices. In the latter case product recovery is limited to biogas and anaerobic sludge.

The effect of product selling price of recovered resources on the TLC is highest for the UASB-DW-RBC and the MBR technology with a factor 1.3 difference between the high and low selling prices (Figure A4.9 in Appendix Chapter 4). For the Conventional Activated Sludge +3R and Aerobic Granular Sludge +3R cases this difference is a factor 1.2. Finally, additional resource recovery for off-site technologies (Conventional Activated Sludge with N, P, Aerobic Granular Sludge and MBR), always results in a lower TLC (worst case scenario 1% difference; best case scenario ~20% difference) than scenarios without resource recovery. Of these three technologies the AGS + 3R is the most financially attractive system.

Variation in pollution removal rates will also impact consumption and production values (Table 4.5) and the corresponding monetary operational costs and benefits (Figure 4.3) and TLC (Figure 4.2). Thus, a 10% lower removal of COD in a UASB will directly affect potential biogas production (also by approximately 10%) and consequently potential revenues. Reported variations in removal efficiencies of UASB systems, struvite precipitation or duckweed production range from, respectively, 55-75% BOD (Chernicharo, 2006), 60-90% P (Le Corre et al., 2009) and 6-13 g dry matter/m^2/d (El-Shafai et al., 2007; Bal Krishna & Polprasert, 2008) and correspond with an increase or decrease of 15-30%. Within this paper, the effect of varying resource selling prices from 0.5-2 (factor 4) was analyzed (Figure A4.9 in the Appendix Chapter 4) and showed a maximum impact of 1.3 on the TLC of the treatment. A 15-30% increase or decrease in the production of recoverable resources (with a similar selling price) is far lower than this factor 4 increase and therefore will impact the TLC far less (< 5%).

Besides variations as a result of varying removal efficiencies, land and recovered resource selling prices, the TLC is impacted by variations in investment costs. Considering (1) the large variations in uncertainty (30%) and the large fraction of capital costs on TLC (Figure 4.2), this impact is considerably higher than the other impacts (see Table A4.8 and A4.9 in the Appendix Chapter 4). However, because these variations impact all systems in an equal matter (e.g. the sewer costs), these price variations will not influence the outcome of our system comparison.

4.3.2 Municipal Solid Waste systems

4.3.2.1 Process performances

The landfill area required for a 3R based system is reduced by 64% in comparison to conventional landfilling (Table 4.6). Because of a lower applied waste generation rate in rural areas for non-organic components, paper and plastic recovery is smaller than in urban areas.

Table 4.6 Calculated resource production and consumption data of the selected MSW system, distinguishing conventional (no resource recovery) and resource recovery (3R) systems in urban and rural areas

Scenario / Parameter	Unit	Conventional		3R decentralized		Centralized 3R	
		Rural	Urban	Rural	Urban	Compost	Dig.& Comp.
Compost	kg/cap/y	-	-	38.9	38.9	39.5	39.5
Paper + Plastics	kg/cap/y	-	-	37.6	48.4	48.4	48.4
Electricity [a]	kWh/cap/y	-	-	-	-	-	17.6
Area collection	m^2/cap	0.00	0.01	0.00	0.01	0.01	0.01
Area 3R	m^2/cap	-	-	0.20	0.20	0.02	0.02
Landfill area	m^2/cap	0.53	0.58	0.18	0.21	0.21	0.21

[a] Electricity production from biogas was calculated using a generator with 40% efficiency (Van Nieuwenhuizen et al., 2011)

Table 4.6 further shows that the per capita energy production from organic waste digestion is approximately equal to the per capita demand of energy of the Aerobic Granular Sludge wastewater treatment system. This offers opportunities for an energy neutral combined wastewater and solid waste treatment system or even an energy producing system when applying the UASB-RBD-DW. Co-digestion of sewage sludge and OSWF has shown the potential to increase biogas production and reactor performance compared to separate digestion of sewage sludge or organic waste (Mata-Alvares et al., 2000; Zupančič et al., 2008; Zitomer et al., 2008), though quality of digested OSWF, with respect to heavy metal content, will decrease (Kujawa-Roeleveld & Zeeman, 2006).

4.3.2.2 Financial analysis of MSW systems

Figure 4.4 shows the per capita TLC after 20 years of operation. The TLC was subdivided in CAPEX at the start, CAPEX at a later stage for landfill extension (using Formula 1) and net operational costs. Figures A4.10 and A4.11 in Section 9 of the Appendix Chapter 4 show, respectively, the numeric values of the TLC and a detailed breakdown for capital expenditures (collection, landfill, 3R facilities and associated land).

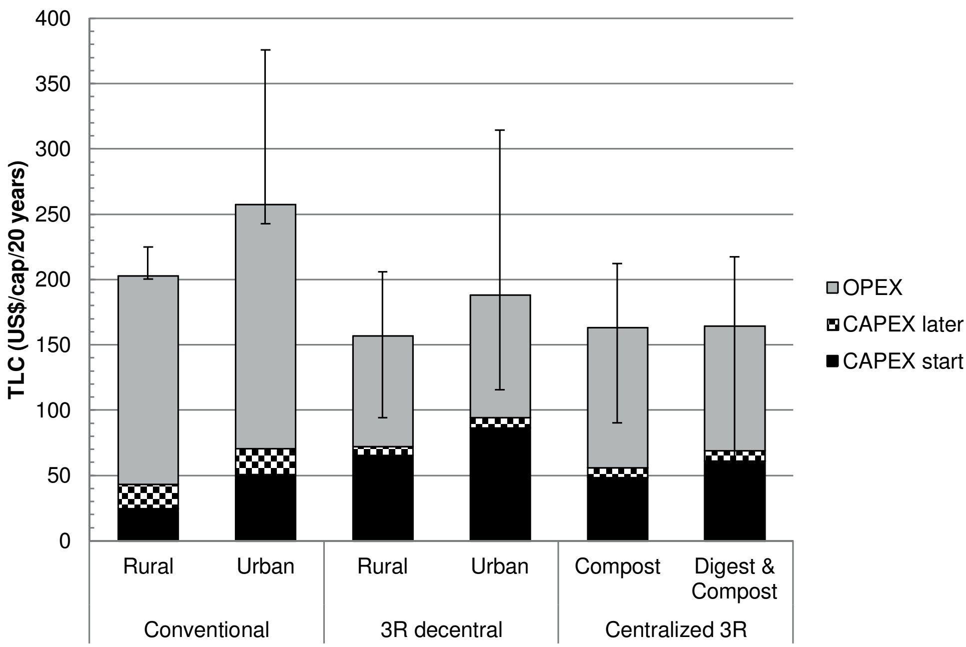

Figure 4.4 Per capita MSW total lifecycle costs (TLC) with distinction between OPEX, CAPEX at start and CAPEX in a later stage. Error bars show maximum and minimum values of variations in land prices (factors 0.5 to 5) and recovered resource selling price (factors 0.5 to 2)

Figure 4.4 shows that a conventional system has lower investments (CAPEX) than a decentralized 3R system. Figure A4.11 in Section 9 of the Appendix Chapter 4shows that nearly 20% of urban decentralized 3R investments are land costs. The applied open windrow composting has low investment and operation costs and requires minimum process control, but

has large area requirement (Veeken, 2005). Land availability, together with labour and wage systems and capability of composters were identified as barriers for development of communal 3R systems (Aprilia et al., 2012). The centralized 3R system, applying only composting, requires less investment than combined digestion and composting 3R systems, but both systems still require lower investment costs than a conventional system. However, the total sum of initial investments (excluding landfill extensions) for the centralized 3R systems is on average higher than those of a conventional system. High total investments in later stages for the conventional system are the result of landfill extensions, which for urban conventional and urban 3R systems system amount to, respectively 20 US$/cap and 8 US$/cap (discounted values) (Figure 4.4).
Figure 4.5 shows per capita operational costs for collection and disposal and net benefits of 3R systems (benefits – costs; presented as negative values). Collection costs of a decentralized 3R system are 20% lower than the other systems, as less waste needs to be transported to a landfill. The highest benefits are calculated for the centralized 3R system, since besides compost, plastic and paper revenues, further revenues are expected from the sale of electricity (biogas).

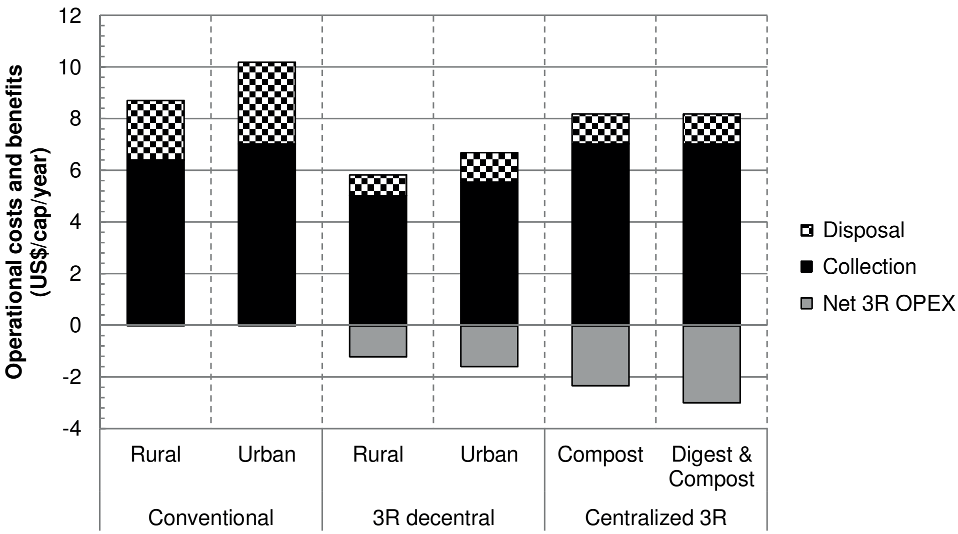

Figure 4.5 Per capita disposal and collection operational costs and net benefits of 3R (shown as negative values) per type of MSW system (US$/cap/y)

Centralized 3R stations have the lowest TLC (Figure 4.4). The decentralized urban 3R system has a 1.15 factor higher TLC but is still more favorable than the conventional system. Several studies on Indonesian solid waste describe costs and benefits of different MSW systems and also show a preference for resource recycling (Aye & Widjaya, 2006; Aprilia et al., 2012). However, these studies include only parts of the MSW system, compare different scales, and apply prices that are too high in the Indonesian context. In addition, the referenced studies include benefits

that are excluded in the current study, such as revenues from the trading of carbon credits under the Kyoto protocol through the Clean Development Mechanism (Aprilia & Tezuka, 2010).
Figure 4.4 further shows the maximum and minimum variations in TLC following variations in land prices (0.5 to 5) and product selling prices (0.5 to 2). Figure A4.13 in the Appendix Chapter 4 shows the impact of each modelled land cost and product selling price, in which the “reference values” are calculated using the values presented in Table A4.4 of the Appendix Chapter 4. The impact of higher or lower recovered product selling prices is much higher than price variations on TLC of wastewater technologies. This is because (1) investment costs of municipal solid waste (Figure 4.4) systems are lower than those for wastewater (Table A4.6 and A4.7 of the Appendix Chapter 4) and (2) resource recovery (e.g. compost and biogas) potential from solid waste (Table 4.6) is higher than for wastewater (Table 4.5). This is most noticeable for the centralized digester and composter, where the difference is almost a factor 4. In this study we have assumed a conservative compost selling price of 400 US$/ton (Table A4.4 of the Appendix Chapter 4), whereas other studies have used compost selling prices of more than double the applied values in this study (Aprilia et al., 2012). The importance of the impact of selling prices shows the need for a proper market analysis of recoverable products in the selection of the type of MSW system.
The effect of variations in land price is highest on the urban decentralized 3R stations (see also Figure A4.11 of the Appendix Chapter 4) and shows a difference of factor 1.8 between the highest and lowest analyzed land prices, whereas this difference is a factor 1.55 for the conventional (urban) systems (Figure A4.13 in the Appendix Chapter 4). The large effect of land price on decentralized (community level) 3R stations calls into question the applicability of these systems in high dense (space scarce) areas.
Variations in organic solid waste compost and biogas production rates or recovery rates of plastics and papers rates lead to variations in the presented specific per capita production rates (Table 4.6) and consequently the TLC. For example, a 20% lower biogas, compost and recoverable plastic and paper production rate will increase the TLC of the “centralized 3R digestion and compost” scenario from 165 to 186 US$/cap/20 years, which equates to a 1.13 factor increase.

4.3.3 Impact of WWT and MSW technical and financial performance on system selection

In Indonesia septic tanks are the minimum treatment requirement (BPS, 2014). Compared to off-site systems, on-site systems and CBS have the lowest TLC, but show the lowest removal efficiencies that cannot comply with standards applicable in (some) urban areas. Evidence suggests that there is a causal link between population density and urban function where a lot of people interact (e.g. cinemas, shopping malls or Commercial Business Districts) and the occurrence of diarrhea (Lasut et al., 2008; Gondhalekar et al., 2013). In areas with high population densities and different urban functions the environment receives higher pollution loads and one person's lack of sanitation can affect the health of many people (Mara et al., 2010). Off-site options should be considered when on-site treatment could entail direct risks to public health or groundwater contamination, or when the risk exists of fecal contamination or eutrophication of surface waters, as is the case in more densely populated areas (UNEP, 2004). In rural areas or low density urban areas the minimum standard (on-site systems) with the lowest costs is recommended. In areas with increasing population densities (e.g. peri-urban areas or higher density rural areas) CBS are proposed, as these show better performance than on-site systems but still have considerably lower costs than off-site systems.

The type of off-site technology selected depends on the local regulations, the land availability and the available funding. In urban areas with high population densities, high land prices and a demand for high water quality, such as in Jakarta or Surabaya (JICA, 2012; Navastara & Navitas, 2012; KNI, 2014) AGS or MBR can be considered, although the latter has nearly 80% higher TLC. In urban areas outside Java with lower land costs anaerobic filters or aerated lagoons can be considered. Resource recovery can contribute to a more attractive TLC, provided recovered or produced products can be sold at a certain price, as is currently being investigated (WSP, 2013b).

Recycling and recovery of urban solid waste by both the formal and informal sector has been applied for many years (Damanhuri & Padmi, 2000), but lacks policies/strategies and financial support. Low involvement of the private sector, inefficiency, and low community awareness led to a low level of service of municipal waste management in Indonesia (Meidiana & Gamse, 2010). The legal framework for municipal solid waste management (Law no. 18/2008) directs that waste generation must be minimized from the source and requires involvement of communities (MoPW, 2013e). Implementation of decentralized 3R systems has been key in MoPW's solid waste policy (Bappenas, 2011). Current success rate of urban communal 3R stations was estimated as 30% due to poor community management (MoPW, 2014b). Consequently, MoPW has started the development of centralized 3R stations. Besides the (potential) improved technical operation, the TLC shows that a shift from decentralized to centralized 3R is financially attractive in urban areas. Continuation of 3R promotion has land use advantages (Table 4.6), results in the recovery of resources and has financial advantages. Local reuse of recovered products (of decentralized 3R systems) will prevent unnecessary transport. Product transport costs were not included in the present study. For urbanized areas with limited space and limited direct reuse potential for recovered resources (e.g. compost) a centralized 3R system is preferred. A market analysis on the demand for recoverable resources is strongly recommended for the selection and location of 3R systems.

The results presented in this study were typically derived on the basis of individually obtained and verifiable input parameters (e.g. specific wastewater production, unit costs and conversion factors). Due to the 'not yet mature' wastewater and solid waste infrastructure in Indonesia, the compiled results cannot be directly verified in the field. With the planned investments and facilities to be constructed, further improvement and verification of results will be possible. However, the applied analysis on varying removal efficiencies, land prices, resource selling prices and investments allows already for identification of uncertainties in the results of TLC.

The selection of wastewater and solid waste systems and their related costs are an important element in planning sanitation systems in developing countries (Parkinson et al., 2014). This study showed how this system selection can be determined based on local conditions (effluent requirements, urban/rural features, land availability, potential sale of recoverable resource). The selection approach for a WWT or MSW system based on residential features can be used for planning purposes not only for Indonesia, but for developing countries in general. The typical performance and per capita resource (e.g. energy, sludge, space) consumption and production data are considered representative for similar (tropical) developing countries (e.g. South East Asia, South America). Following a cost update on investments and operation unit costs parameters, the presented approach can be applied for wastewater and municipal solid waste system selection in other developing countries facing similar challenges as Indonesia.

Besides population density and urban features, applicability of systems is determined by other factors as well. Application of septic tanks depends also on ground water levels, soil conditions and availability of a (piped) water supply (Loetscher & Keller, 2002). Similarly, CBS systems have been successfully applied in densely populated areas (Eales et al., 2013). Community Sanitation Centers, where people come for bathing, washing and toilet use (Ulrich et al., 2009) were excluded from this study. These systems are now less promoted since more and more people have their own water supply at home and are often no longer applied (Eales et al., 2013).

In this analysis the practical suitability of off-site processes was not quantified, but should be included in the final system selection. In more remote areas skilled labour, spare parts or required energy and chemicals may be not available and systems depending on them are not appropriate, whereas simpler (e.g. anaerobic systems, lagoons or duckweed ponds) systems are (Senzia et al., 2003). Disregarding requirements for operation and maintenance in system selection will ultimately result in system failure. Besides the impact this may have on public health and the environment, failure also represent a loss of investment.

Existing sewer systems are of poor quality (USAID, 2006) and in the design and construction of new systems improvements are needed. Furthermore, specific demands by a municipality or property developer (JICA, 2012; KNI, 2014) or wishes of a foreign donor (Aprilia & Tezuka, 2010) play a role in system selection of both WWT and MSW systems. Moreover, successful operation of facilities depends on the institutional capacity of the responsible actors. Despite increasing attention, this still needs further improvement and must be considered in the system selection (Kearton et al., 2013; USDP, 2014). In this study “new sanitation” systems (Zeeman & Kujawa-Roeleveld, 2011) were excluded, as the required household level source separation and corresponding transport were considered too “high tech” in the current Indonesian setting. Demonstrations could be considered for specific small scale greenfield application.

4.4 Conclusions

The study showed how the feasibility of wastewater and solid waste systems for developing countries can be analyzed on the basis of technical and financial performance. The technical performance provided insight into wastewater treatment efficiencies as well as wastewater and solid waste per capita resource production and consumption (land use, phosphorus, energy, duckweed, compost, water, plastics and papers) parameters in the Indonesian context. By putting a monetary value to investments, land costs, technical operational and recoverable resources the financial features of each system could be determined. A combination of technical and financial performance was used to recommend different systems for different residential scenarios. This confirms our hypothesis that the selection of feasible wastewater or solid waste systems can be based on a few key (and typically available) parameters like urban/rural features and land availability (related to residential densities). The cost estimates can be used for the assessment of required wastewater and solid waste investment and operational budgets in the planning process. An analysis whereby land prices and revenues from recovered resources were varied, showed that the effect of high land prices on land intensive WWT systems have a major impact (nearly a factor 5), whereas the effect on TLC by varying selling prices of recovered resources showed a maximum factor of 1.3 difference. For MSW, variations in land price impacted the TLC up to a factor of 1.8, while variation of the selling prices of recovered resources impacted the TLC by a factor 4.

The technical analysis was based on scientific literature and case studies from developing countries. Following a cost update on investments and operation unit cost parameters, the presented approach can be applied in other developing countries. The presented results and conclusions allow decision support for Indonesia and can be considered as guidelines and orientation for the implementation of improved wastewater and solid waste concepts in other developing countries.

Acknowledgments

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Appendix Chapter 4

Section 1 Applied wastewater production values

Table A4.1 Applied wastewater production values

Parameter	Unit	Production features		
		Total	Grey water	Black water
Flow	l/cap/d	136	100	36
COD[a]	g/cap/d	82	34	48
BOD[b]	g/cap/d	41	17	24
Total Nitrogen (TN) [a]	g/cap/d	12	0	12
Total Phosphorus (TP) [a]	g/cap/d	2	0.2	1.8

[a] Derived from Almy (2008), [b] a COD/BOD ratio of 2 is applied following Meinzinger & Oldenburg (2009)

Section 2 Basis for WWT process performance in this study

Septic tank: Presented removal efficiencies in Table 4.2 of Chapter 4 are of black water and are based on average reported literature values (Chernicharo 2006; Van der Graaf et al. 1989; Ulrich et al. 2009; Lettinga et al. 1991; Van Voorthuizen et al. 2008). A sludge production of 35 l/person/year is applied (WSP 2013). Based on the removal efficiency, a black water COD/TSS (Total Suspended Solids) of 1 (Meinzinger & Oldenburg 2009) and a biodegradability (76%) and methanogenesis rate (86%) (Halalsheh 2002) a sludge production of 3.5 kg TSS/cap/year is calculated. This results in a TSS content of 10%, which is much higher than the TS content of 1.5% measured at the sludge treatment facility (JICA 2012). This is attributed to the way septic tanks are emptied by typically sucking out the complete tank (WSP 2013). Pathogen removal for septic tanks and other technologies are based on Tchobanoglous et al. (2003).
ABR+ AF and Anaerobic Filter: Performance is based on Ulrich et al. (2009) and verified with other sources (Reynaud et al. 2012a; Said 2000; Wibisono et al. 2003). Sludge production follows the approach described for septic tanks, in which 68% of incoming total COD is suspended (Halalsheh 2002). The Anaerobic Filter (Said 2000; Kearton et al. 2013) follows the process conditions of the ABR+AF, but sludge treatment takes place at the location of the WWTP (e.g. sludge drying beds or mechanical thickening) (MoPW 2013b).
Aerated lagoons and Conventional Activated Sludge systems (CAS): Removal efficiencies, sludge production and energy requirement of the CAS are determined using a model developed by Royal HaskoningDHV that calculates sludge production based on Chudoba & Ferdinand (1985), oxygen consumption using Beuthe (1970), N removal using EPA (1993), P removal using Janssen et al. (2002). The applied sludge load is 0.31 kg COD/kg MLSS.d at a temperature of 29ºC (DIY_PU 2010). Removal efficiencies in the aerated lagoon are the same as CAS, but

sludge production, HRT and loading are based on MoPW's guideline (MoPW 2013b), whereas energy consumption are based on EPA (2002).

<u>CAS + N&P and AGS</u>: The CAS_N&P applies anaerobic-aerobic conditions for P-removal (Baetens 2001) and a pre-denitrification zone for enhanced N removal, following the Phoredox set-up (Brett et al. 1997). The Aerobic Granular Sludge (AGS) process, is a compact, low energy consuming SBR system, applying fast settling aerobic granules, but with a higher P content in the sludge (De Kreuk et al. 2005; De Kreuk et al. 2007). The design of a compact installation is based on the possibility of simultaneous nitrification/denitrification (SND) within the granules. De Kreuk et al. (2005) reports high removal efficiencies for phosphate (94%) and total nitrogen (94%). Calculations are based on the same models as CAS. Applied sludge loading is 0.27 kg COD/kg MLSS.d. Additional $FeCl_3$ dosing was modeled to meet a value of maximum 5 mg/l to comply with the irrigation standard (MoPW 2001).

<u>MBR</u>: The MBR process set-up is similar to the CAS N&P, but membranes with 100% TSS retention instead of clarifiers are applied to separate sludge from treated effluent. The MBR is based on *Zenon ZW 500D* submerged membranes (design flux 33 $l/m^2/h$) and energy consumption of the membrane system is calculated using Van Bentem et al. (2006).

<u>Additional resource recovery technologies for CAS N&P, AGS and MBR</u>: Resource recovery in the CAS N&P, AGS and MBR can be applied by modifications in the sludge line. Recoverable resources comprise (1) energy, applying sludge digestion (2) struvite by precipitation of phosphate in the centrate of dewatered digested sludge and (3) composting of sludge.

- Energy: Sludge reduction and biogas production in a CSTR sludge digester with 20 days HRT is calculated using Chen and Hashimoto (Chen & Hashimoto 1980) with a typical organic fraction degradation of 40% (Van Nieuwenhuizen et al. 2011). Gas production is based on 0.35 m^3 CH_4/kg COD (STP) and 1.42 g COD/g VS (Droste 1997).
- Struvite: P-release during the digestion follows organic sludge degradation (Bi et al. 2013; Ju et al. 1999), whereas P release from precipitated $FePO_4(H_2O)_2$ following reduction of Fe^{3+} to Fe^{2+} is determined as 50% using the "OliAnalyzer" software package. Selected P-recovery method in this study is struvite crystallization, using a Crystalactor® reactor which has been particularly successful from high P concentrated flows (Cornel & Schaum 2009; Giesen 1999). Reported P-removal efficiencies approach 90% or more provided a sufficient high pH of about 9 is applied (Wu et al. 2001; Le Corre et al. 2009; Battistoni et al. 2002). In this study a P-recovery of 88% is selected, applying an over dosage of 2 mmol Mg^{2+}.
- Compost: Composting is described in in the municipal solid waste section.

<u>UASB-DW-RBC</u>: The technology comprises 3 units.

1. UASB (Lettinga et al. 1980; Seghezzo et al. 1998; Lettinga et al. 1993). COD and BOD removal efficiencies are based on Draaijer et al. (1992) and verified with other sources reported values (Chernicharo 2006; Lettinga et al. 1991; de Graaff et al. 2011). N, P removal are based on Van Voorthuizen et al. (2008), thus resulting in a 74%, 75%, 15% and 5%

removal of, respectively, COD, BOD, TN and TP. Biogas production was calculated based on a COD balance: $COD_{in} = COD_{eff} + COD_{Sludge} + COD_{sulphate} + COD_{biogas}$, (Draaijer et al. 1992).

- COD lost to sulphate reduction is 12% (Draaijer et al. 1992).
- COD of sludge was calculated using the specific sludge production of Draaijer et al. (1992) of 0.4 kg TSS/kg TSS_{in} and a COD/TSS ratio of 1.06 as reported by Halalsheh (2002).The applied TS/COD ratio of the influent was defined as 0.35, based on a 0.3 value reported by Halalsheh (2002), and grey water and black water ratio of, respectively, 0.3 by and 0.44 reported by Lettinga et al. (1991) for Indonesian wastewater, thus COD in sludge equals 15% of the influent COD.
- Using a 74% COD removal, the effluent COD is 0.26 of COD in the influent.
- the fraction of influent COD converted to biogas becomes 1-0.12-0.15-0.26=0.47.

2. Duckweed ponds (DW) aim to polish the UASB effluent for N, P and produce duckweed. Process conditions and duckweed production were determined as shown in Table A4.2. N and P removal efficiencies were calculated based on N and P first order kinetics coefficients (K in d^{-1}) determined in a batch process by Korner & Vermaat (1998). N and P effluent values in a continuous system were calculated as a function of influent N and P values, the constants (K) and the Hydraulic Retention Time (HRT), using the standard formula A4.1 for the CSTR (Tchobanoglous et al. 2003):

$$N, P_{effluent\ conc} = \frac{N, P_{influent\ conc}}{(1 + K * HRT)} \qquad \text{(Formula A4.1)}$$

Besides N, P removal by duckweed other processes take place, like nitrification, denitrification and precipitation (Korner & Vermaat 1998; Al-Nozaily et al. 2000; Vermaat & Khalid Hanif 1998) and removal by duckweed was calculated as shown in Table A4.2. Verification of applied method for N and P removal was done by recalculating the values in the studies of El-Shafai et al. (2007) and Bal Krishna & Polprasert (2008) using the K values by Korner & Vermaat (1998). Thus, El-Shafai et al. (2007) measured Duckweed N and P–uptake rates of, respectively, 0.44 g $N/m^2.d$ and 0.09 g $P/m^2.d$, compared to recalculated values of 0.49 g $N/m^2.d$ and 0.07 g $P/m^2.d$. Bal Krishna & Polprasert (2008) determined N uptake rates of 0.62 g $N/m^2.d$ compared to a recalculated value of 0.75 g $N/m^2.d$. Calculated DW N and P removal efficiencies in this study were 86% and 63%, respectively.

Table A4.2 Approach for duckweed pond, process calculations

	Parameters to determine	Selected value or equation	Applied references; (letters A-F) refer to indicated parameters
A	Duckweed Yields	10 g dry weight/m^2.d	(El-Shafai et al. 2007) [A,B,C,F, H], (Bal Krishna & Polprasert 2008) [A,B, F], (Alaerts et al. 1996) [A, F], (Oron et al. 1987) [A], (Korner & Vermaat 1998)[D, E, F, G], (Al-Nozaily et al. 2000) [A, F, G], (Culley et al. 1973) [A], (Vermaat & Khalid Hanif 1998)[F, G]
B	Organic Loading rate	50 kg COD/ha.d	
C	Area (A)	A = HRT x Q/Depth, HRT=10 days, Depth = 0.48 m	
D	Total N-removal coefficient K	0.41 d^{-1} based on 73 mg/l N	
E	Total P-removal coefficient K	0.18 d^{-1} based on 14 mg/l P	
F	% N-rem. by DW uptake	70% of total N-removal	
G	% P-rem.by DW uptake	60% of total P-removal	
H	COD and BOD removal	COD: 64%, BOD 73%	

3. The RBC is operated as an OLAND system (Oxygen-limited autotrophic nitrification/ denitrification) (De Clippeleir et al. 2011; Windey et al. 2005) and aims to reduce the final nitrogen levels. The system features low sludge production, energy consumption, area requirement and N_2O emissions (Mulder 2003). COD and BOD removal was based on literature (Vlaeminck et al. 2009; Friedler 2004; Tervahauta et al. 2013), while N removal was based on De Clippeleir et al. (2011), applying an N-Kjeldahl removal of 89%, a 22% NO_3-N production of removed NKj, and a N-total removal of 51%. Sludge production follows the MoPW guideline (MoPW 2013b). Energy consumption averages reported values (Vlaeminck et al. 2009; MoPW 2013b; Van Buuren 2010; Cortez et al. 2008; Trang 2011).

Three example mass balances are shown in Figure A4.1 (septic tank), Figure A4.2 (MBR without resource recovery), Figure A4.3 (AGS + resource recovery).

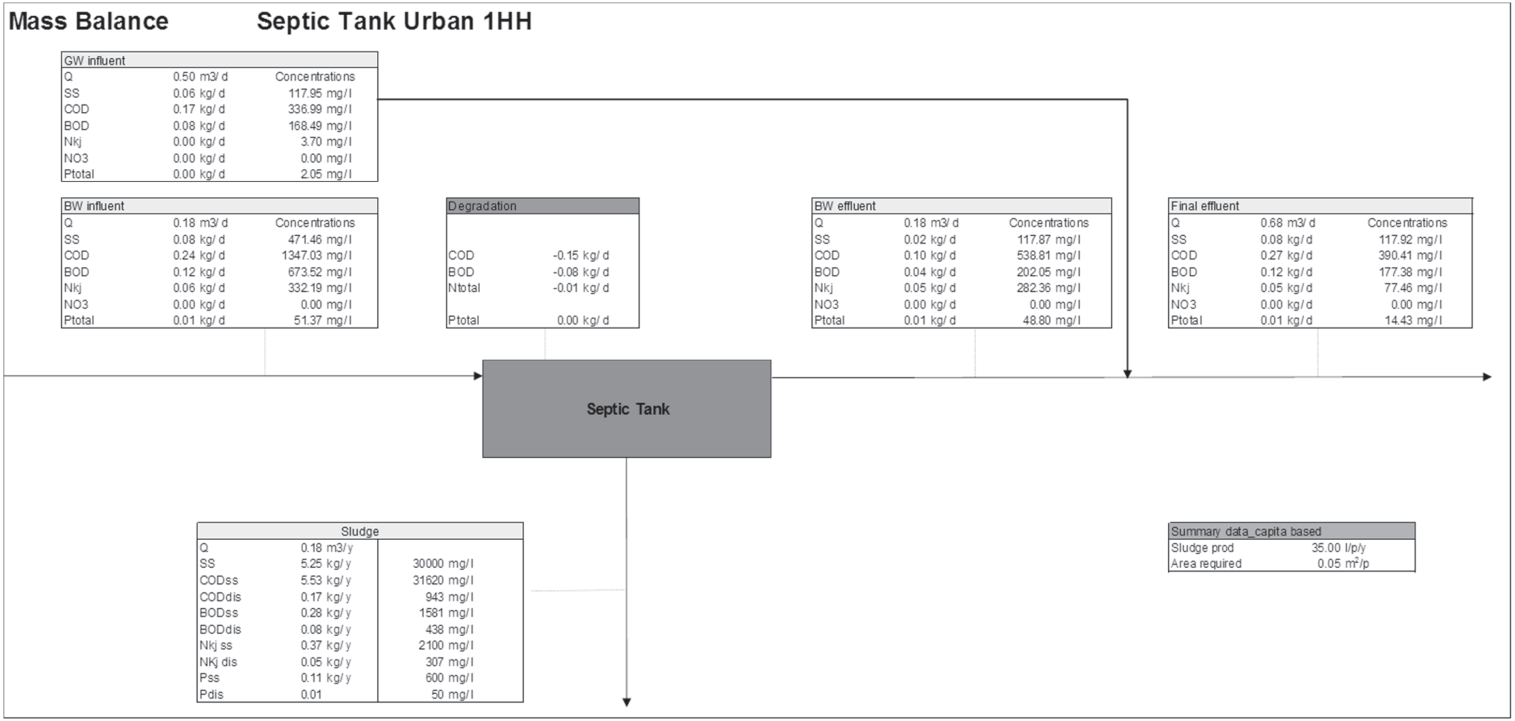


Figure A4.1 Mass balance of on-site system (septic tank) for 1 HH (household)

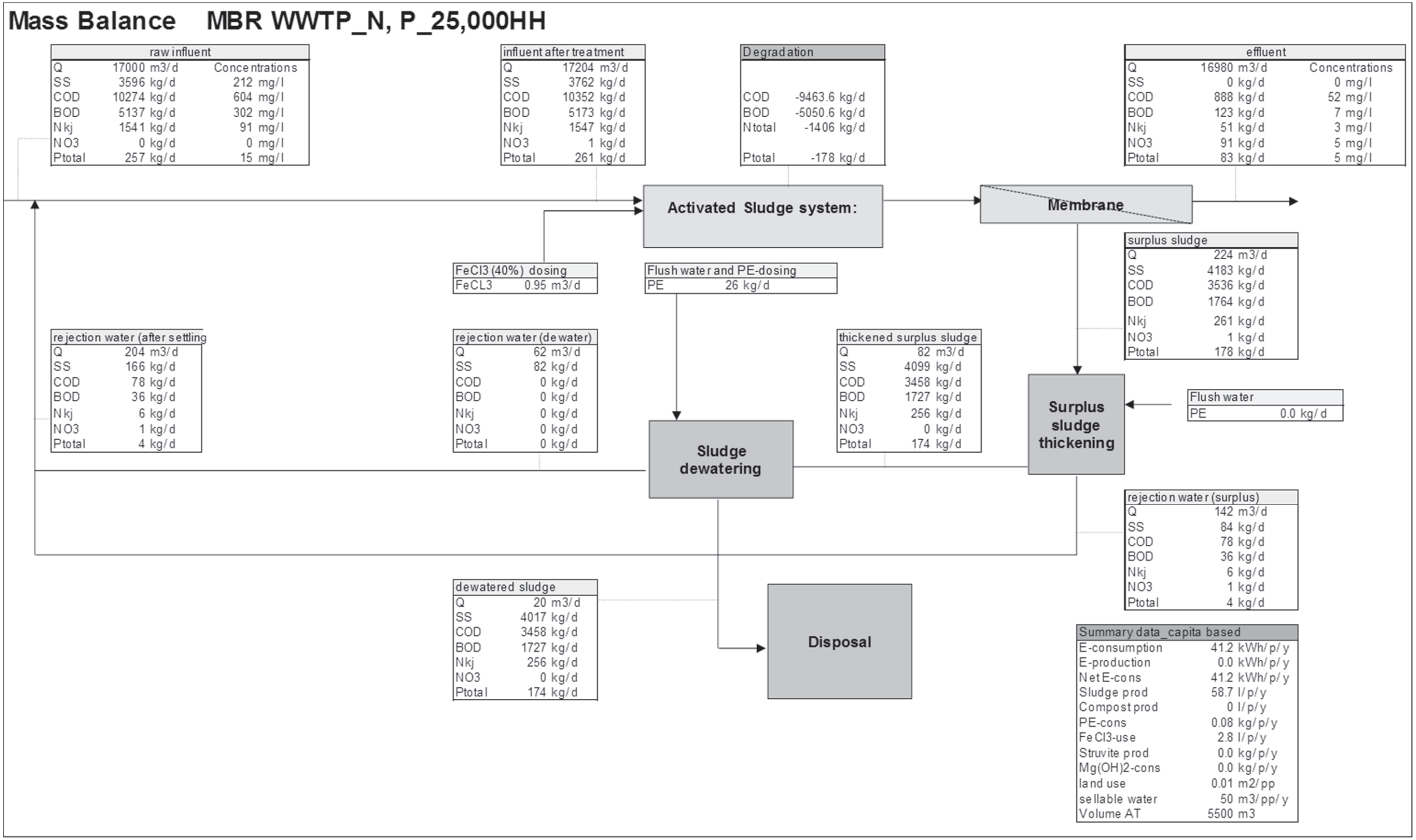


Figure A4.2 Mass balance of MBR scenario for 25,000 HH (households) without resource recovery

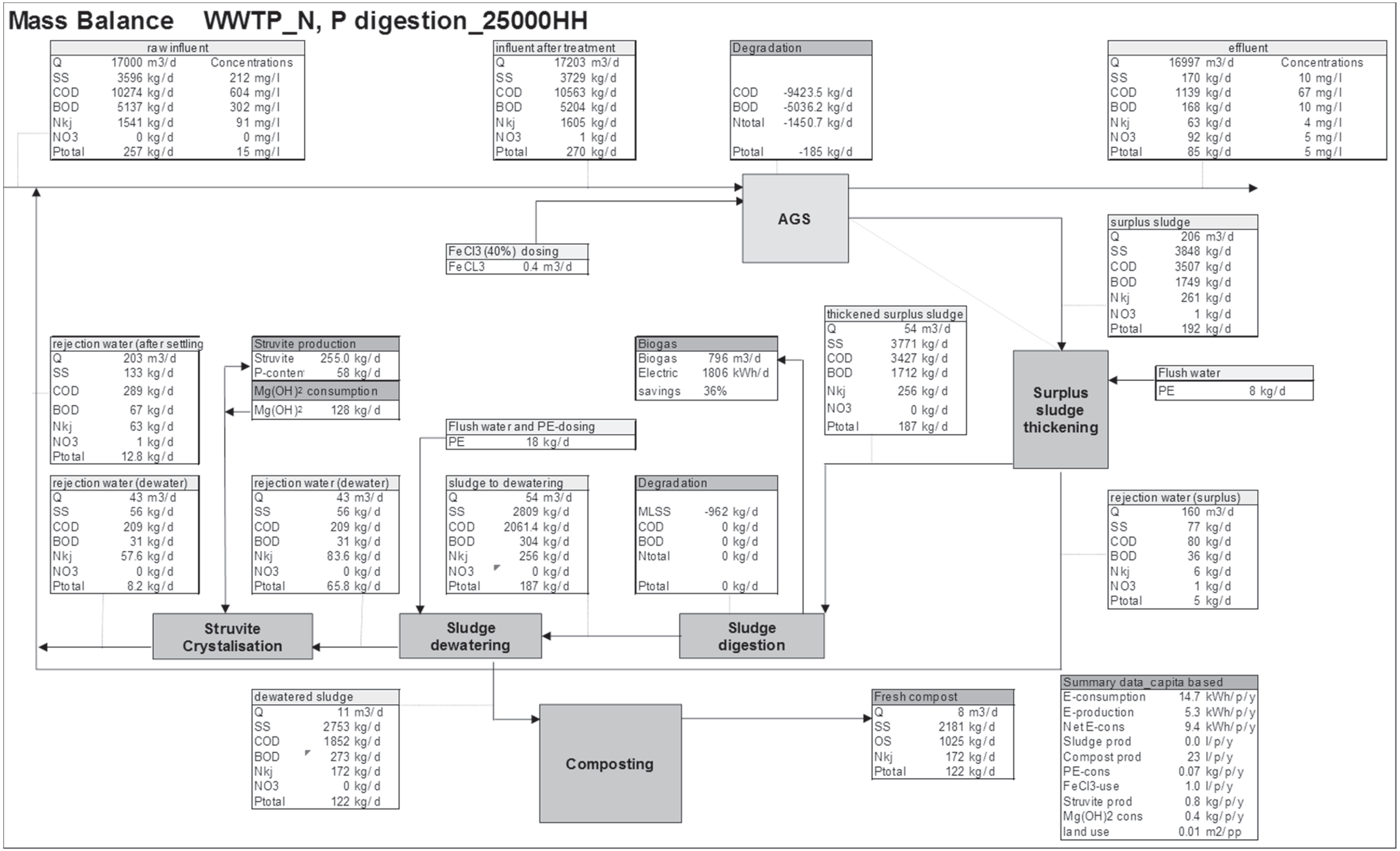


Figure A4.3 Mass balance of AGS scenario for 25,000 HH (households) with sludge digestion, struvite crystallization and composting

Section 3 Methodology to determine costs for off-site sewer systems

Costs for simplified and pumped sanitary systems for Medium Centralized and Centralized systems are based on Loetscher described in Van Buuren (2010). These are then verified with reported design values in Indonesia by the IndII program (SKM 2011a; SKM 2011b) and constructed systems in Banjarmasin (PDPAL-Banjarmasin 2013). The applied formula for calculation was:

$C_{cHD} = 5.04*X*G*T* (D^{-0.35})*(5610*(H-10)^{-0.46} + 1800)$ (Van Buuren 2010; see formula 7.11)

Where:

C_{cHD} = construction costs per household depending on population density (USD/household);

D = population density in persons/ha;

H = number of households connected (> 10 households);

G = dimensionless factor that expresses the impact of soil nature on costs;

T = dimensionless factor that expresses impact of traffic impediment on costs;

X = dimensionless factor that expresses the relative capital costs of different sewerage types such as conventional sewerage, simplified sewerage, covered drains and settled sewerage.

In the analysis the applied sewer costs for Medium Centralized systems are the average of small scale and mid-scale systems, whereas the applied sewer costs for Centralized systems are the average of large scale and city systems from

Table A4.3 Applied parameters for determination of off-site sewer system costs following Van Buuren (2010)

Sewer system features									
Para-meters	System	Medium Centralized				Centralized			
		small scale		mid-scale		large scale		City scale	
	Description	Range	Selected	Range	Selected	Range	Selected	Range	Selected
H	Households connected	200-2000	1,000	2,000-5,000	5,000	5,000-15,000	10,000	15,000-50,000	25,000
D	Density (pp/ha) [a]	100-175	138	175-250	213	>250	275	>250	300
G	Impact of soil nature on costs	1-1.6	1.3	1-1.6	1.3	1-1.6	1.3	1-1.6	1.3
T	Impact of traffic impediment on costs	1-1.33	1	1-1.33	1.11	1-1.33	1.22	1-1.33	1.33
X	Relative capital costs of sewer type	0.43-1	0.43	0.43-1	0.59		0.75		0.75
Selected sewer type		simplified small		simplified small		pumped sanitary		pumped sanitary	
Sewer costs US$		1,020,000		6,300,000		15,800,000		41,250,000	
Pump costs US$		9,300		26,900		42,800		98,300	
Total costs US$		1,029,300		6,326,900		15,842,800		41,348,300	
Investments per household of 5 people(US$)		1,029		1,265		1,584		1,654	
Investments per person (US$)		210		250		320		330	

[a] (TTPS 2009)

In “green field” situations, where sewer construction can be combined with other development, a reduction of 50% of sewer construction costs is applied following the costs breakdown specified in Rioned (2007)

Figure A4.4 shows the comparison of calculated, designed, constructed and greenfield sewer system investment cost presented in Indonesian Rp (10,000 Rp equals 1 US$).

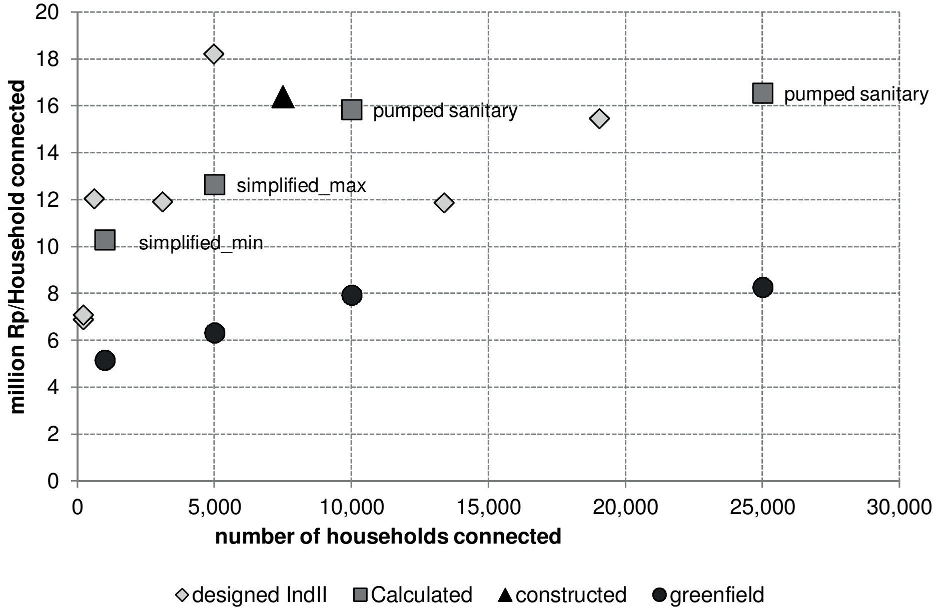


Figure A4.4 Sewer system costs per household (in Rp; where 10,000 Rp equals 1 US$) comparison of calculated sewer costs (Table A4.3) and IndII program (SKM 2011a; SKM 2011b) and constructed systems in Banjarmasin (PDPAL-Banjarmasin 2013) and greenfield situation (Rioned 2007)

Section 4 Basis for MSW process performance in this study

Additional justification on applied approach is presented below

- Collection of waste: Purity of recovered products (especially organics) can be increased using at source separation (Saveyn & Eder 2014). Aprilia et al. (2012) shows that less than 20% of the households of a monitored community in Jakarta apply source separation. In the current study all solid waste is assumed to be mixed and separated at decentral or central 3R stations with recovery percentages presented in Table 4.3 of Chapter 4;
- Plastic and paper recovery: Recovered plastic is cleaned and compressed. Paper is bound together. Products are collected by “agents” of processing companies (Banda Aceh IPLT 2013);

- Composting: During composting OSWF is oxidized to CO_2 and H_2O, while producing heat that evaporates water and sanitises the compost. The common practice in Indonesia is to use open windrows (MoPW 2013a), which was assumed in this analysis. Composting balances are based on Veeken et al. (2002) and Veeken et al. (2003). The applied heat production is 20 MJ/kg $OS_{converted}$ (Haug 1993), TS content is 40% (Hamelers 2001; Bhattacharya et al. 2005), VS/TS ratio is 65% (Zhu et al. 2010; Zitomer et al. 2008; Norbu et al. 2005) and the maximum biodegradability is 65% (Attero 2014). The calculated value of 0.35 kg compost/kg Organic waste is the same as reported by PU (2013a) for small scale composters. Compost has a density of 0.65 kg/l (Veeken et al. 2005);
- Digestion: The OSWF introduced into the digester is determined as 63%. Thus, sufficient heat production potential is maintained in the OSWF that is send to the composter directly to evaporate water in the total (fresh and digested) OSWF to come to a final desired TS content of 65% (Hamelers 2001). In case a bigger fraction of the OSWF is digested the final compost becomes too wet and additional dewatering and leachate treatment is required. If a smaller fraction is applied less biogas is produced. A 40% electricity conversion efficiency for biogas engines is used (Van Nieuwenhuizen et al. 2011);
- Other technologies: Incineration of waste would result in a volume reduction, heat and electricity production and production of recoverable materials, but is not found feasible in non-OECD countries, due to cost and frequently unsuitable waste composition (Aprilia et al. 2012).

The mass balances are shown in Figure A4.5 (urban conventional), Figure A4.6 (decentral 3R), Figure A4.7 (central 3R with composting) and Figure A4.8 (central 3R with digestion and composting). Mass balances are based on 200,000 people, as this is the typical mean size of a municipality in Indonesia (50% percentile) corrected for low density (< 25 pp/ha) rural areas (BPS 2014). These low density rural areas were excluded, as an MSW collection system is not found feasible for these remote areas (MoPW 2014).

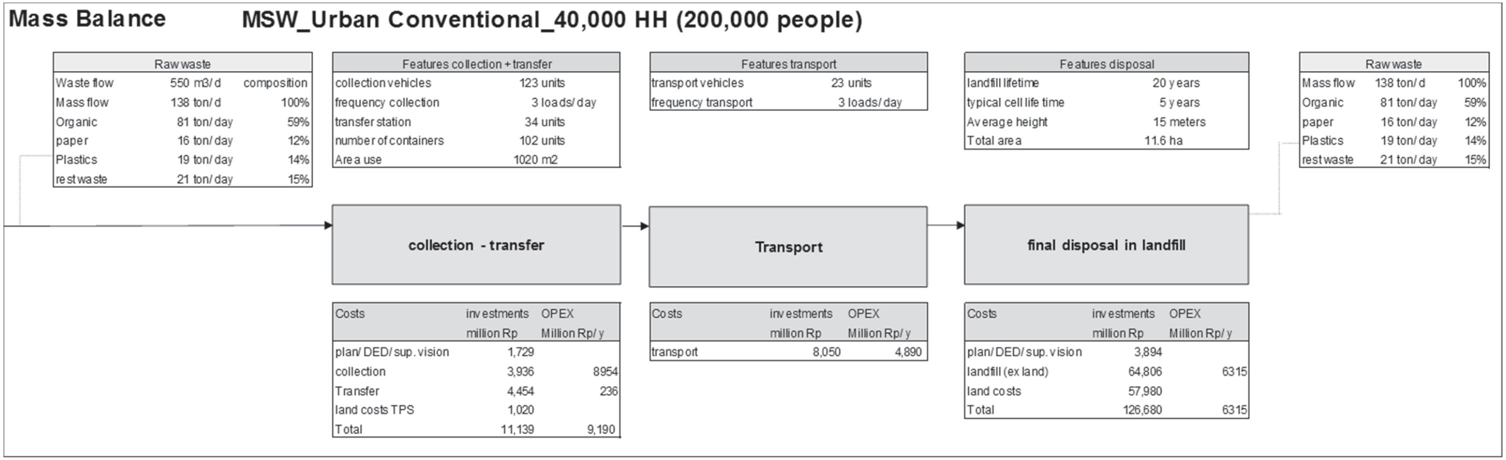


Figure A4.5 Mass balance of the MSW urban conventional system for 40,000 HH (households). Indicated prices are in Indonesian Rupiah (Rp); 10,000 Rp = 1 US$

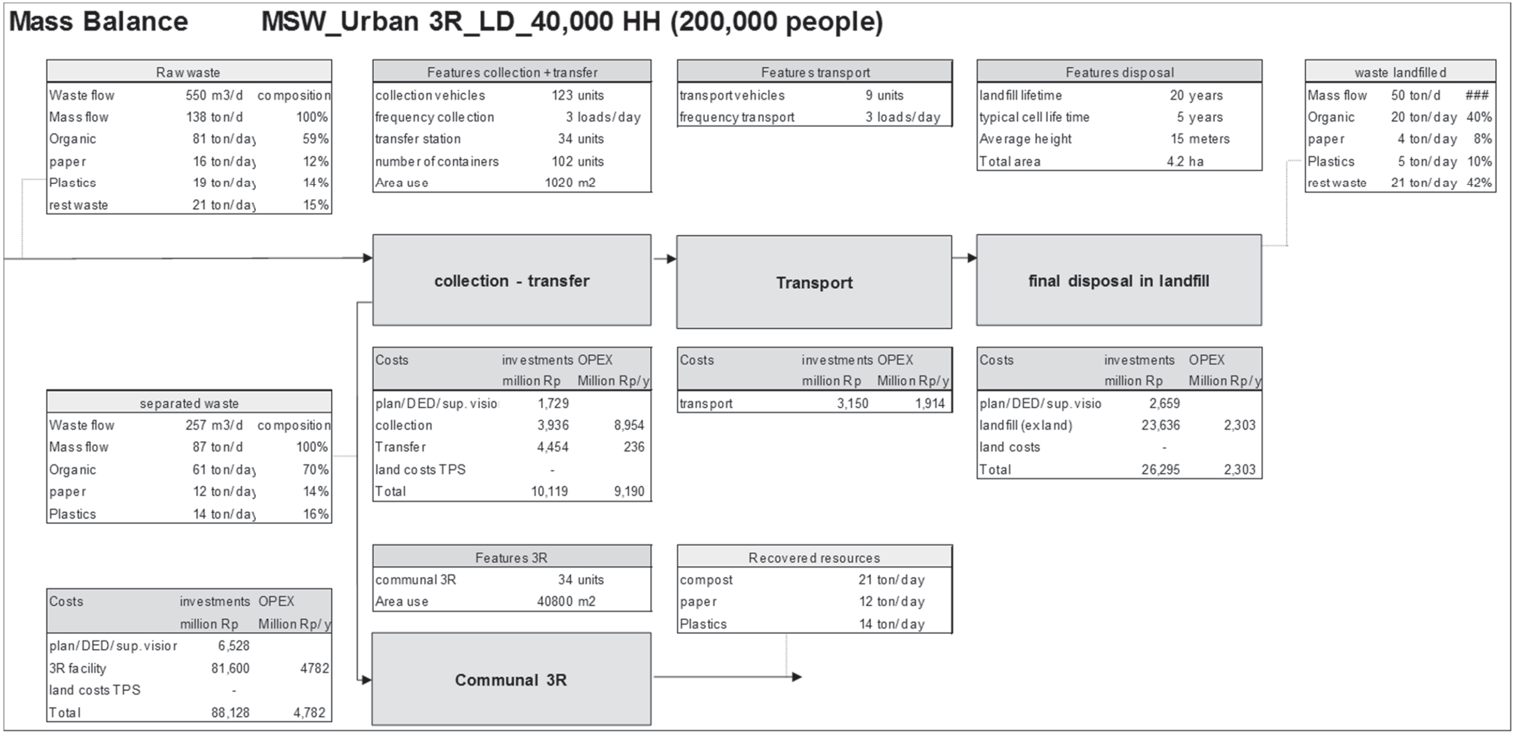


Figure A4.6 Mass balance of the MSW urban low density (LD) 3R decentralized system for 40,000 HH (households). Indicated prices are in Indonesian Rupiah (Rp); 10,000 Rp = 1 US$

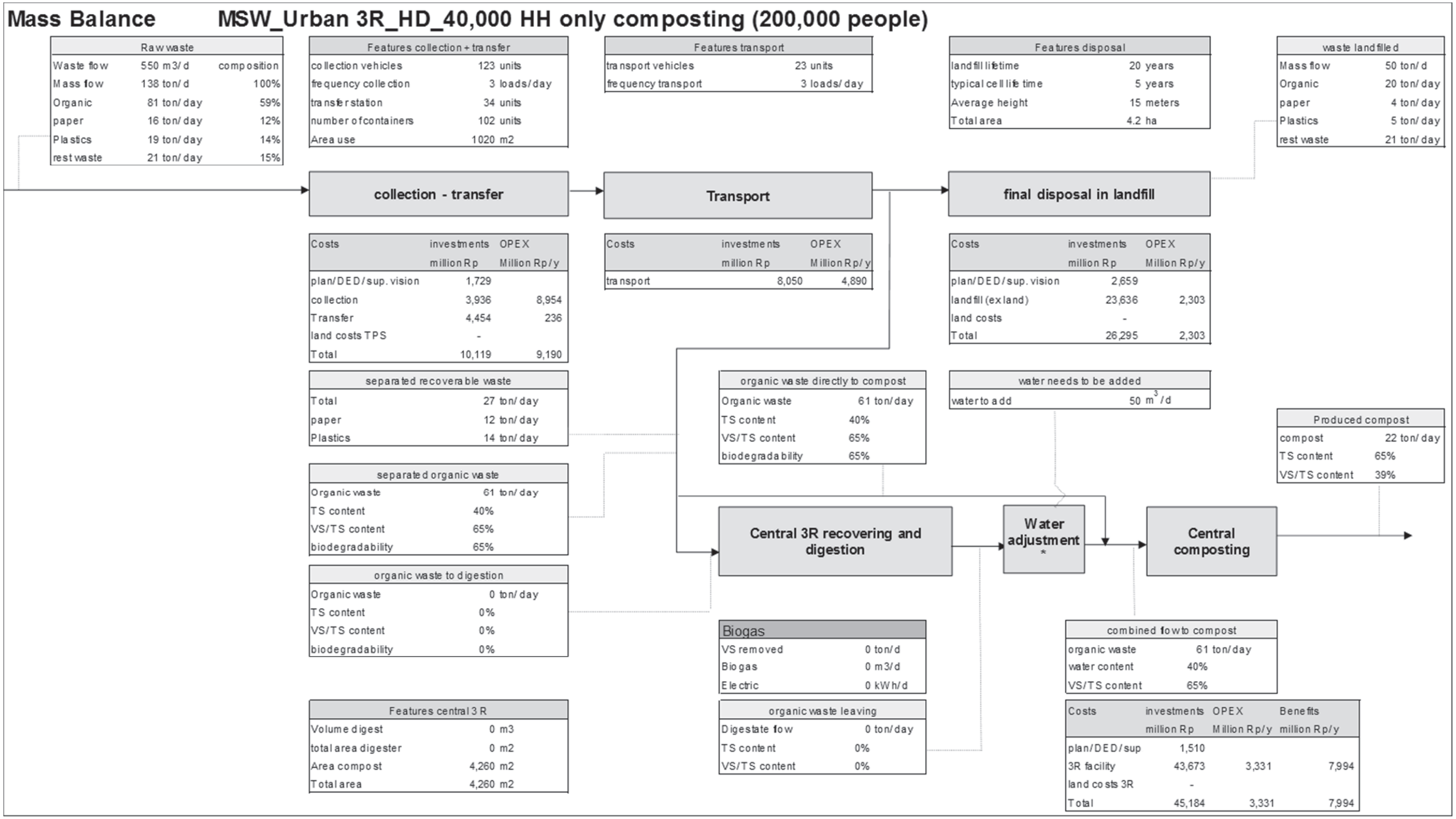


Figure A4.7 Mass balance of the MSW urban high density (HD) 3R centralized composting only system for 40,000 HH (households). Indicated prices are in Indonesian Rupiah (Rp); 10,000 Rp = 1 US$

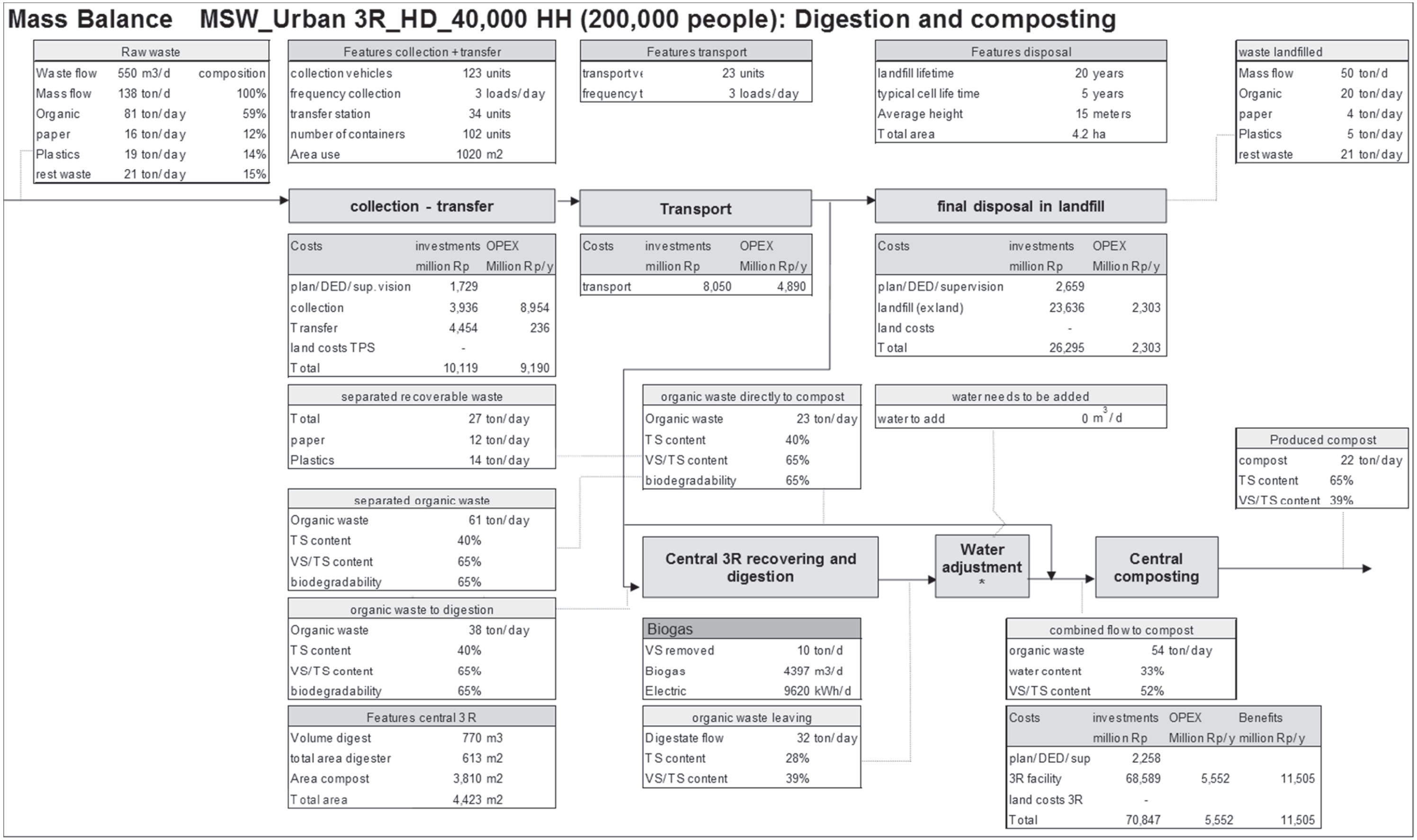


Figure A4.8 Mass balance of the MSW urban high density (HD) 3R centralized digestion and composting system for 40,000 HH (households). Indicated prices are in Indonesian Rupiah (Rp); 10,000 Rp = 1 US$

Section 5 Applied specific operational cost and benefit prices in this study

Table A4.4 Applied specific operational cost and benefit prices of indicated parameters

Parameter	Price Unit	Source
Sludge disposal or collection	15 US$/ton	At the IPLT Banda Aceh and Pulo Gebang prices of 15 US$ for 1-2 m^3 are applied; private and industrial companies are charged 8-30 US$/ton for disposal (Banda Aceh IPLT 2013), (Pulo Gebang 2014).
40% $FeCl_3$ solution	200 US$/t	Based on reference quotation from Weifang Menjie Chemicals Co., Ltd (Menjie 2014)
PE (sludge thickening)	4 US$/kg	Supply price to industries (FFI 2014)
Electricity	0.1 US$/kWh	Municipal users pay 0.04-0.14 US$/kWh depending on connection. Applied value corresponds with 3,500-5,500 VA. (MoEMR 2012)
Paper, Plastic selling price	50 US$/t	Field data (Banda Aceh IPLT 2013) show prices of up to 200 US$/ton depending on the quality. In the sensitivity analysis the effect of selling price is determined.
Compost selling price	30 US$/t of sludge	Compost selling prices from cow manure and OSWF were as high as 200 US$/ton (BiRu 2013), (ITB compost facility 2014) but also free supply to farmers is applied (Pulo Gebang 2014). The effect of price variations is studied in a sensitivity analysis.
	40 US$/t of OSWF	
Struvite selling price	975 US$/t	Based on reference quotation from Wuhan Xingzhengshun Import & Export Co., Ltd (XingZhenshun 2014)
MBR effluent	0.05 US$/$m^3$	Based on bulk price of surface water in Jababeka, Cikarang estate (Cikarang Estate 2014)
Duckweed selling price	100 US$/t fresh duckweed	Reported selling prices of fresh duckweed are as high as 200-300 US$/ton (Journey et al. 1993; El-Shafai et al. 2007). In this study 100 US$/ton is used and effect of price variations are studied in a sensitivity analysis.
Land Prices	10 US$/$m^2$	For rural landfills estimated based on Bappeda (2014)
	50 US$/$m^2$	WWT infrastructure, landfill developments near urban areas and developments of MSW collection activities in rural residential areas (Bappeda 2014; Navastara & Navitas 2012).
	100 US$/$m^2$	Developments of MSW collection activities in urban residential areas (Bappeda 2014; Navastara & Navitas 2012).
	Because land prices fluctuate considerably (Navastara & Navitas 2012), the effect of land price is analysed in a sensitivity analysis.	

Dutch price levels for CAS, AGS and MBR systems were based on designs prepared by Royal HaskoningDHV. These were converted to Indonesian prices based on discussions with Dutch and Indonesian engineers using price levels in Indonesia compared to the Netherlands of 60%, 80% and 80% for respectively civil, electrical and mechanical components.

Section 6 Energy Consumption (Cons), Production (Prod), Net use and recoverable fraction WWT technical performance

Table A4.5 Energy Consumption (Cons), Production (Prod), Net use and recoverable fraction

parameter	Unit	An.Fil.	Aer. Lag.	CAS	CAS N, P	AGS	MBR	UASB-DW-RBC	CAS +3R	AGS +3R	MBR +3R
Energy Cons	kWh/cap/y	1.6	11.9	22.9	23.6	14.0	41.2	4.3	24.4	14.7	41.7
Energy Prod	kWh/cap/y	0.0	0.0	0.0	0.0	0.0	0.0	19.6	5.3	5.3	5.3
Net Energy	kWh/cap/y	1.6	11.9	22.9	23.6	14.0	41.2	-15.3	19.1	9.4	36.6
Recovered	%	0	0	0	0	0	0	456	22	36	16

Section 7 Negative impact associated with the application of digested sludge

(1) Health, (2) social acceptability and (3) environmental complications are associated with the application of digested sludge (Tesfamariam et al. 2013; Koné et al. 2007):

1. To minimize health risks resulting from helminthes ova (Jimenez-Cisneros 2008), a composting process of at least 2 months is suggested (Koné et al. 2007), although 99% fecal coliforms reduction can be expected after 10 days of composting (Way 2013). To reduce pathogen content the heat production potential of digested sludge in the composting process can be increased by addition of (non-digested) waste activated sludge as long as permeability is in an optimum range (Veeken et al. 2003). Alternatively, co-composting of digested sewage sludge with OSWF has shown benefits in pathogens removal (Koné et al. 2007; Strauss et al. 1997).
2. Social acceptability of (fecal) sludge products must be considered. Starkl et al. (2010) describe reluctance in use of by-products from human feces, despite (potential) financial benefits. Similar resistance was found during a visit to a biogas project for individual households, where digesters were equipped with influent pipes for cow manure and black water from toilets. Only 1% of 800 households still applied the black water connection, as compost was expected to be less attractive for buyers (BiRu 2013).
3. Environmental concerns are related to N and P levels reaching environmentally toxic levels and has caused governing authorities to set limits to how much sludge can be applied to agronomic land (Tesfamariam et al. 2013). Heavy metals, although not specified in the Indonesian compost standard (BSN 2004), could be another barrier in the application of processed sludge. Sewage sludge compost generally meets the proposed (European Union) limit values for Cd, Cr, Hg, Ni, Pb and Zn but tends to have problems in meeting the proposed Cu limits (Saveyn & Eder 2014).

Section 8 Detailed overview of calculated WWT investment and OPEX

Table A4.6 Investment (sewer, treatment technology and land costs), yearly OPEX (costs, benefits and net OPEX) and total lifecycle costs (TLC) [a] for on-site, CBS and MedCen systems

	System	Unit	on-site	CBS	Off-site: medium centralized										
	Technology		Septic + IPLT	ABR + AF	An. Fil.	Aerated lagoon	CAS	CAS + N, P	AGS	MBR	UASB-DW-RBC	CAS+ 3R	AGS+ 3R	MBR+ 3R
Investments	*Sewer*	*US$/cap*	*0*	*114*	*229*	*229*	*229*	*229*	*229*	*229*	*229*	*229*	*229*	*229*
	Treatment		*100*	*64*	*43*	*28*	*142*	*178*	*171*	*324*	*135*	*195*	*188*	*341*
	Land		*0*	*10*	*4*	*15*	*2*	*3*	*1*	*1*	*236*	*3*	*2*	*2*
	Total		100	188	276	272	374	410	401	555	601	427	419	572
Yearly OPEX	*Costs*	*US$/cap/y*	*2.5*	*3.1*	*5.1*	*6.5*	*9.2*	*10.5*	*9.1*	*15.7*	*6.3*	*10.3*	*9.0*	*15.5*
	Benefits		*0.1*	*0.1*	*0.0*	*0.0*	*0.0*	*0.0*	*0.0*	*2.5*	*3.6*	*2.0*	*1.8*	*4.4*
	Net OPEX		2.4	3.0	5.1	6.5	9.2	10.5	9.1	13.2	2.7	8.3	7.2	11.1
TLC 20 years		US$/cap	**143**	**243**	**369**	**391**	**542**	**602**	**568**	**796**	**650**	**580**	**551**	**775**

[a] Calculated based on Formula 1 in Chapter 4

Table A4.7 Investment (sewer, treatment and land costs), yearly OPEX (Costs, benefits and net OPEX) and total lifecycle costs (TLC) [a] for Centralized systems

	System	Unit	Off-site: centralized									
	Technology		An.Fil.	Aerated lagoon	CAS	CAS + N, P	AGS	MBR	UASB-DW-RBC	CAS+ 3R	AGS+ 3R	MBR+ 3R
Investments	*Sewer*	*US$/cap*	*324*	*324*	*324*	*324*	*324*	*324*	*324*	*324*	*324*	*324*
	Treatment		*33*	*28*	*86*	*107*	*103*	*196*	*82*	*116*	*112*	*204*
	Land		*4*	*15*	*2*	*3*	*1*	*1*	*236*	*3*	*2*	*2*
	Total		361	367	412	434	428	521	642	442	437	530
Yearly OPEX	*Costs*	*US$/cap/y*	*3.4*	*4.6*	*6.8*	*7.8*	*6.4*	*12.2*	*4.1*	*7.3*	*5.9*	*11.7*
	Benefits		*0.0*	*0.0*	*0.0*	*0.0*	*0.0*	*2.5*	*3.6*	*2.0*	*1.8*	*4.4*
	Net OPEX		3.4	4.6	6.8	7.8	6.4	9.8	0.5	5.3	4.1	7.2
TLC 20 years		US$/cap	**423**	**452**	**537**	**577**	**546**	**700**	**650**	**539**	**512**	**662**

[a] Calculated based on Formula 1 in Chapter 4

Table A4.8 Minimum and maximum Investment (sewer, treatment technology and land costs), yearly OPEX (costs, benefits and net OPEX) and total lifecycle costs (TLC) [a] for on-site, CBS and MedCen systems with a 30% range on sewer and treatment investment costs

	System	Unit	on-site	CBS	Off-site: medium centralized									
	Technology		Septic + IPLT	ABR + AF	An. Fil.	Aerated lagoon	CAS	CAS + N, P	AGS	MBR	UASB-DW-RBC	CAS+ 3R	AGS+ 3R	MBR+ 3R
Investments	*Sewer*	*US$/cap*	*0 - 0*	*80 - 148*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*	*161 - 298*
	Treatment		*70 - 130*	*45 - 83*	*30 - 55*	*20 - 37*	*99 - 185*	*124 - 231*	*119 - 222*	*227 - 421*	*95 - 176*	*137 - 254*	*132 - 244*	*239 - 444*
	Land		*0*	*10*	*4*	*15*	*2*	*3*	*1*	*1*	*236*	*3*	*2*	*2*
	Total		*70 - 130*	*135 - 241*	*195 - 358*	*195 - 350*	*262 - 485*	*288 - 532*	*281 - 521*	*389 - 721*	*491 - 710*	*300 - 555*	*294 - 544*	*401 - 744*
Yearly OPEX	*Costs*	*US$/cap/y*	*2.5*	*3.1*	*5.1*	*6.5*	*9.2*	*10.5*	*9.1*	*15.7*	*6.3*	*10.3*	*9.0*	*15.5*
	Benefits		*0.1*	*0.1*	*0.0*	*0.0*	*0.0*	*0.0*	*0.0*	*2.5*	*3.6*	*2.0*	*1.8*	*4.4*
	Net OPEX		*2.4*	*3.0*	*5.1*	*6.5*	*9.2*	*10.5*	*9.1*	*13.2*	*2.7*	*8.3*	*7.2*	*11.1*
TLC 20 years		US$/cap	113 - 173	189 - 296	288 - 451	314 - 469	431 - 654	480 - 724	448 - 688	630 - 962	541 - 760	452 - 707	425 - 676	604 - 946

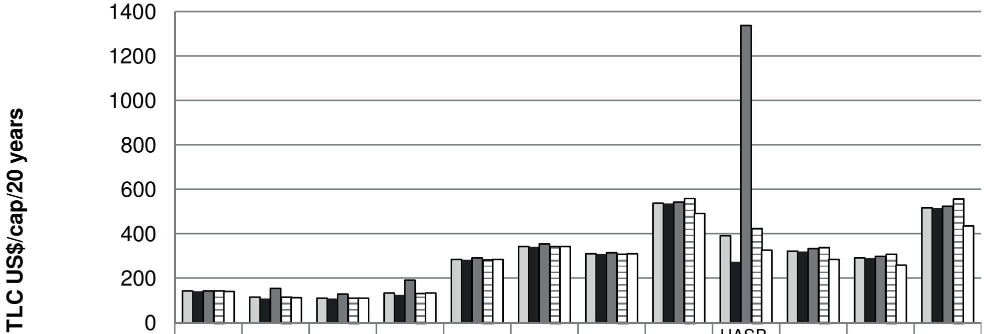
[a] Calculated based on Formula 1 in Chapter 4

Table A4.9 Maximum and minimum Investment (sewer, treatment technology and land costs), yearly OPEX (costs, benefits and net OPEX) and total lifecycle costs (TLC) [a] for Centralized systems with a 30% range on sewer and treatment investment costs

	System	Unit	Off-site: centralized									
	Technology		An. Fil.	Aerated lagoon	CAS	CAS + N, P	AGS	MBR	UASB-DW-RBC	CAS+ 3R	AGS+ 3R	MBR+ 3R
Investments	Sewer	US$/cap	227 - 421	227 - 421	227 - 421	227 - 421	227 - 421	227 - 421	227 - 421	227 - 421	227 - 421	227 - 421
	Treatment		23 - 42	20 - 37	60 - 112	75 - 140	72 - 134	137 - 255	57 - 106	81 - 151	78 - 145	143 - 265
	Land		4	15	2	3	1	1	236	3	2	2
	Total		254 - 468	261 - 472	289 - 535	305 - 563	300 - 556	365 - 677	520 - 763	311 - 574	306 - 567	371 - 688
Yearly OPEX	Costs	US$/cap/y	3.4	4.6	6.8	7.8	6.4	12.2	4.1	7.3	5.9	11.7
	Benefits		0.0	0.0	0.0	0.0	0.0	2.5	3.6	2.0	1.8	4.4
	Net OPEX		3.4	4.6	6.8	7.8	6.4	9.8	0.5	5.3	4.1	7.2
TLC 20 years		US$/cap	316 - 530	346 - 557	414 - 660	448 - 706	418 - 674	544 - 856	528 - 772	407 - 671	382 - 643	504 - 820

[a] Calculated based on Formula 1 in Chapter 4

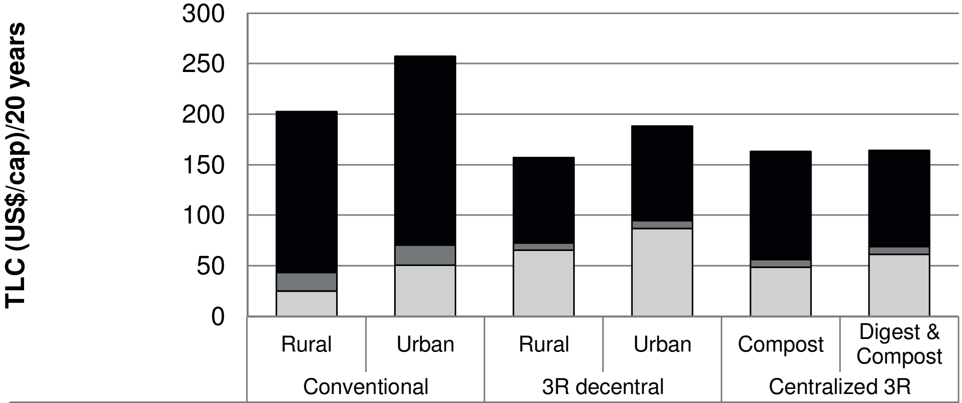
An impact analysis (Figure A4.9) is performed based on reference prices (Table A4.5 of this Appendix) with land prices being varied from 0.5 to 5 times the reference values. Product selling prices were varied from 0.5 to 2 times the reference values. As sewer system costs are the same for all medium centralized systems, the TLC shown in Figure A4.9 only include the investment and operational costs and benefits for land acquisition and treatment systems. The maximum and minimum differences with the reference price were included in Figure 4.2 of Chapter 4.



	septic+I PLT	ABR + AF	An.Fil.	aer. Lagoon	CAS	CAS + N, P	AGS	MBR	UASB-DW-RBC	CAS+ 3R	AGS+ 3R	MBR+ 3R
	on-site	CBS	Med Cen									
Reference US$/cap	143	116	112	134	285	345	311	539	393	322	293	518
0.5 X LC US$/cap	143	111	110	127	284	343	310	538	275	321	292	517
5 x LC US$/cap	144	156	129	193	293	355	316	544	1338	333	299	524
0.5 X Prod US$/cap	145	118	112	134	285	345	311	561	426	341	309	558
2 x Prod US$/cap	141	114	112	134	285	345	311	493	327	286	261	436

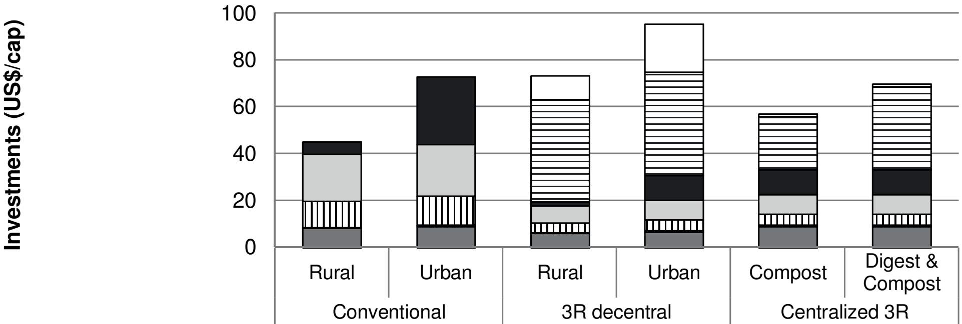
Figure A4.9 Per capita WWT TLC Sensitivity analysis for land costs (LC) and recovered product (Prod) prices for technologies excluding sewer investment and operational costs

Section 9 Financial analysis of MSW systems



	Conventional		3R decentral		Centralized 3R	
	Rural	Urban	Rural	Urban	Compost	Digest & Compost
TLC US$/cap	203	258	157	188	164	165
■ OPEX US$/cap	160	187	85	94	107	95
■ CAPEX later US$/cap	18	20	7	8	8	8
□ CAPEX start US$/cap	25	51	66	87	49	61

Figure A4.10 Per capita MSW Total lifecycle Cost (TLC) with distinction between OPEX, CAPEX at start and CAPEX in a later stage



	Conventional		3R decentral		Centralized 3R	
	Rural	Urban	Rural	Urban	Compost	Digest & Compost
□ 3R land US$/cap	0.00	0.00	10.08	20.40	1.07	1.11
□ 3R treatment US$/cap	0.00	0.00	43.52	44.06	22.59	35.42
■ disposal land US$/cap	5.27	28.99	1.83	10.57	10.57	10.57
□ Landfill extensions US$/cap	20.12	22.06	7.40	8.44	8.44	8.44
□ Landfill first phase US$/cap	11.21	12.29	4.12	4.70	4.70	4.70
□ Collection land US$/cap	0.23	0.51	0.23	0.51	0.51	0.51
■ Collection US$/cap	8.34	9.08	6.06	6.63	9.08	9.08

Figure A4.11 Non-discounted per capita investment costs (US$/cap) per type of MSW system, distinguishing land (collection, disposal and treatment) and investments (collection, treatment facilities, initial and extension of landfill

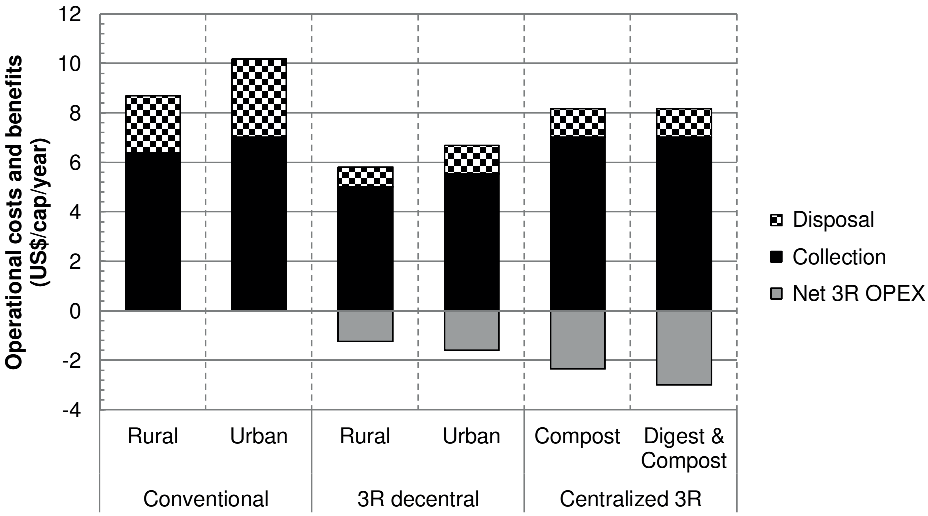
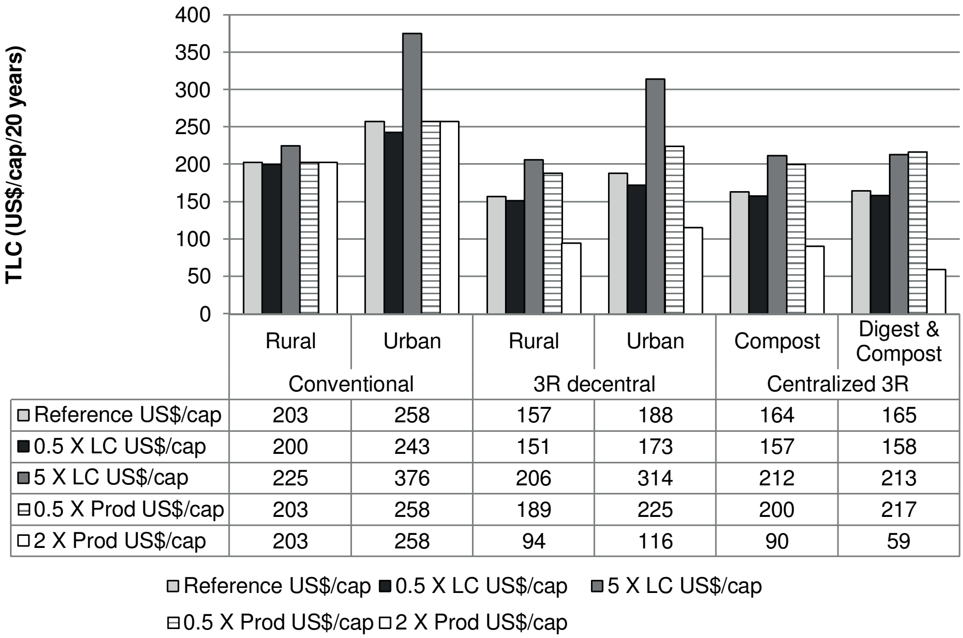


Figure A4.12 Per capita disposal and collection operational costs and net benefits of 3R (shown as negative values) per type of MSW system (US$/cap/y)



	Conventional		3R decentral		Centralized 3R	
	Rural	Urban	Rural	Urban	Compost	Digest & Compost
Reference US$/cap	203	258	157	188	164	165
0.5 X LC US$/cap	200	243	151	173	157	158
5 X LC US$/cap	225	376	206	314	212	213
0.5 X Prod US$/cap	203	258	189	225	200	217
2 X Prod US$/cap	203	258	94	116	90	59

Figure A4.13 Per capita MSW TLC Sensitivity analysis for land costs (LC) and recovered product (Prod) prices

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Chapter 5

A new approach to nationwide sanitation planning for developing countries: case study of Indonesia

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Abstract

Many developing countries struggle to provide wastewater and solid waste services. The backlog in access has been partly attributed to the absence of a functional sanitation planning framework. Various planning tools are available; however a comprehensive framework that directly links a government policy to nationwide planning is missing. Therefore, we propose a framework to facilitate the nationwide planning process for the implementation of wastewater and solid waste services. The framework requires inputs from government planners and experts in the formulation of starting points and targets. Based on a limited number of indicators (population density, urban functions) three outputs are generated. The first output is a visualization of the spatial distribution of wastewater and solid waste systems to support regional priority setting in planning and create awareness. Secondly, the total number of people served, budget requirements and distribution of systems is determined. Thirdly, the required budget is allocated to the responsible institution to assure effective implementation. The determined budgets are specified by their beneficiaries, distinguishing urban, rural, poor and non-poor households. The framework was applied for Indonesia and outputs were adopted in the National Development Plan. The required budget to reach the Indonesian government's 2019 target was determined to be 25 billion US$ over 5 years. The contribution from the national budget required a more than fivefold increase compared to the current budget allocation in Indonesia, corresponding to an increase from 0.5 to 2.7 billion US$ per year. The budget for campaigning, advocacy and institutional strengthening to enable implementation was determined to be 10% of the total budget. The proposed framework is not only suitable for Indonesia, but could also be applied to any developing country that aims to increase access to wastewater and solid waste facilities.

Keywords: wastewater, solid waste, nationwide planning, investment and operational costs, GIS

5.1 Introduction

Between 2005 and 2010, developing Asia experienced remarkably higher annualized growth rates (7.3% in Gross Domestic Product per capita) than other developing regions like Sub Saharan Africa and Latin America and the Caribbean (both 2.3%) (ADB - Asian Development Bank, 2012). However, this rapid economic growth had limited impact on improved access to wastewater facilities which was 55% (ADB, 2012), corresponding with an annual 3% increase since 1990. Solid waste services in Asian cities is about 20% (Hutton et al., 2008) and showed limited increase, comparing Indonesian health data (2010-2013) which showed an increase from 23.4% in 2010 to 24.9% in 2013 only (Ministry of Health, 2010, 2013). The Millennium Development Goals (MDG) aimed to halve the proportion of people without access to wastewater facilities by 2015 compared to 1990. A progress report shows that a number of (South East) Asian countries, such as Indonesia, Cambodia and India, did not meet these targets (WHO & UNICEF, 2015). A challenge for governments to reach the 2015 MDG and the defined 100% access Sustainable Development Goals (SDG) target in 2030 (United Nations, 2015) is the absence of a functional management framework dealing with planning and budgeting (Baum et al., 2013; WHO & UNICEF, 2014).

Sanitation frameworks aim to respond to real needs and make informed decisions about investments for sanitation improvements involving the resources to meet recognized priorities (Törnqvist et al., 2008; Parkinson et al., 2014). Existing sanitation planning frameworks typically focus on specific population groups, distinguishing urban, rural or poor or non-poor communities (Törnqvist et al., 2008; Mehta & Movik, 2010; Sijbesma, 2011), while a comprehensive planning framework that incorporates all these citizens is required. Planning frameworks further differ in level of complexity, ranging from simple methodologies relying on guiding principles and check lists, like the Sanitation 21-framework (Parkinson et al., 2014) to more complex ones, including material flow analysis (MFA) (Meinzinger et al., 2009) or Quantitative Microbial Risk Assessment (Surinkul & Koottatep, 2009). One example for the latter is the SANEX decision support system (Loetscher & Keller, 2002), which consists of several steps, including a selection and screening of feasible technologies on a range of criteria considering settlements characteristics, soil characteristics, quality of water supply, community profiles and pollution control measures. SANEX has been tested in small scale communities in several developing countries, including Indonesia (Loetscher & Keller, 2002). However, the more complex frameworks like SANEX are often budget and time demanding and hardly applicable for a nationwide long term sanitation planning, because required data input, such as soil conditions or quality of water supply are not available on a nationwide level (Törnqvist et al., 2008). The simpler ones may not provide the required level of insight to respond to real needs.

In addition to wastewater, solid waste also contributes to pollution. Therefore, a comprehensive approach addressing both sanitation sub-sectors is desired (Ersoy et al., 2008; WSP - Water and Sanitation Program of the World Bank, 2011; ADB, 2013a). In several sanitation practices, like community lead total sanitation (CLTS) (Mehta & Movik, 2010) (basic) household waste management is considered. However, the CLTS approach focusses on rural areas only. Analytical tools, like MFA, can also include solid waste flows and may support sanitation planning (Meinzinger et al., 2009). MFAs, though, cannot be readily up-scaled for nationwide planning due to their complexity.

An available framework designed for nationwide sanitation planning and budgeting is the Service Delivery Assessment (SDA) of WSP (WSP, 2014). The SDA consists of (1) a review of past sanitation access, (2) a costing model, and (3) a diagnosis of service delivery bottlenecks. It lacks, however, a wastewater system selection based on residential features and neglects the impact of untreated sewage on public health and the environment.

To organize and integrate wastewater and solid waste systems in land use planning activities, and to support regional priority setting and to create awareness of the required implementation, visualization in Geographic Information Systems (GIS) can be used. (Quaye-Ballard & An, 2010; Coutinho-Rodrigues et al., 2011; Gondhalekar et al., 2013). However, sanitation frameworks that present their output in GIS are scarce and are not readily available.

Table 5.1 summarizes the differences between described existing frameworks and our proposed framework. It shows that none of the reviewed frameworks considers all the described elements to develop a sanitation plan from a (national) governmental sanitation (wastewater and solid waste) policy.

In this paper, a new wastewater and solid waste planning framework is presented that directly links government policies to a nationwide planning roadmap. The framework requires input from government planners and experts in the formulation of starting points and targets. It then only requires a number of key indicators to arrive at: (1) spatial planning visualized in GIS, (2) required number of facilities and budgets per population group and (3) allocation of budgets to implementing institutions. The application of the framework is demonstrated using Indonesia as an example.

Table 5.1 Overview of sanitation frameworks and elements considered: target group, sanitation wastewater treatment (WWT) or municipal solid waste (MSW) sub-sector, health and environment (H&E) based selection, application of readily available criteria, budget calculation and or (institutional) allocation, and GIS based output

Sanitation Framework or approach	Target group	Sub-sector focus:	H&E based system selection	Readily available criteria	Budget calculation and/ or allocation	GIS based output	Reference
Planning sustainable water & sanitation systems in peri-urban areas	Peri-urban	WWT	yes	**no**	**no**	**no**	Törnqvist et al., 2008
CLTS	Rural	WWT, MSW	yes	yes	no	no	Mehta & Movik, 2010
Sanitation Financing models for urban poor	Urban-poor	WWT, MSW	**no**	yes	yes	no	Sijbesma, 2011
Sanitation 21-framework	Urban	WWT	yes	yes	**no**	**no**	Parkinson et al., 2014
Material Flow Analysis	Urban/rural	WWT/MSW	yes	no	**no**	**no**	Meinzinger et al., 2009
Quantitative Microbial Risk Assessment	Urban/rural	WWT	yes	**no**	**no**	**no**	Surinkul & Koottatep, 2009
SANEX	Urban/rural	WWT	yes	**no**	yes	**no**	Loetscher & Keller, 2002
Service Delivery Assessment	Urban/rural	WWT	no	yes	yes	**no**	WSP, 2014
GIS based sanitation analysis	Urban/rural	WWT	yes	**no**	**no**	yes	Coutinho-Rodrigues et al., 2011; Gondhalekar et al., 2013
Our framework	Urban/rural; poor /non-poor	WWT, MSW	yes	yes	yes	yes	

The national average of access to wastewater facilities in Indonesia was 56% in 2010, with highest access in the urban areas (73%) (Ministry of Health, 2010). Barely 1% of the population is connected to a sewer system (Kearton et al., 2013). Most of the installed wastewater infrastructure comprises septic tanks. However, 95% of the septic tanks leach liquid directly into the ground or discharge to surface water (WSP, 2013a). The current septage sludge management system is performing poorly in terms of technical and financial operation (WSP, 2013b). Only 25% of the Indonesian population are served by a solid waste management system (Ministry of Health, 2013). The lack of adequate wastewater systems, combined with inadequate solid waste management, is causing contamination of surface and groundwater (ADB, 2013a). Increased government attention towards sanitation resulted in an increased investment from 0.6 to 1.5 billion US$ per year between 2010 and 2014 (USDP - Urban Sanitation Development Program, 2014). In 2013 the Ministry of Planning started preparing the National Medium Term Development Plan (2015-2019) for which the framework presented here was applied. In that plan, the "universal targets" were introduced which define that the entire population must have access to wastewater facilities; 70% of the population should be served by a solid waste management system; and a 20% reduction in landfilling of household waste should be achieved (Bappenas, 2014a).

5.2 Description of the sanitation planning framework

The planning framework aims to generate three policy and planning relevant outputs. Firstly the spatial distribution of selected wastewater and solid waste systems is visualized to support regional priority setting and create awareness of the required implementation. Secondly, the total number of people served, budget requirements and number of systems are determined. A distinction is made between urban, rural as well as poor and non-poor population groups. These population groups have different (i) access to sanitation facilities (WHO & UNICEF, 2014; WSP, 2014), (ii) future access targets (e.g. urban and rural) (Bappenas, 2014a), (iii) implementing agencies (rural implementation typically through the Ministry of health (Mehta & Movik, 2010; ODI - Overseas Development Institute, 2011); urban implementation through the ministry of public works or construction (Yan et al., 2006; WSP, 2014)) and (iv) support needs (e.g. financial needs of poor communities) (Sijbesma, 2011). Thirdly, the required budget is allocated to the responsible institution to assure effective implementation. In the framework, these three domains are addressed in a structured manner and policy and planning relevant outputs are produced (Figure 5.1). Outputs are generated in three process flows, namely (i) inception definition; (ii) data collection, and (iii) data processing. To facilitate the replication of the method, the steps 1-8 are numbered in the order that the framework should be applied.

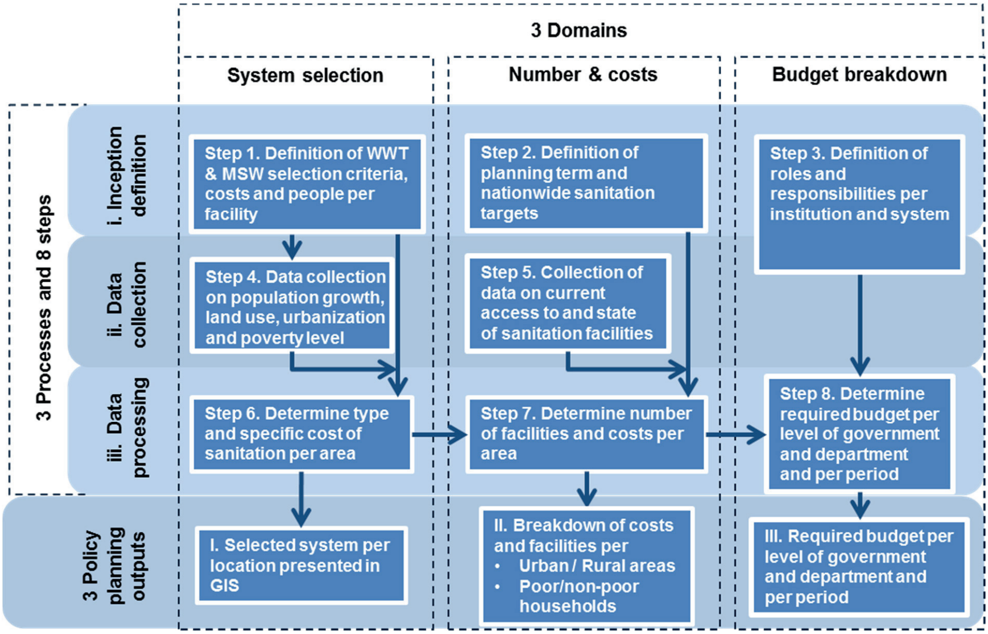

Figure 5.1 Framework for determination of 3 Policy Planning outputs in corresponding domains, distinguishing processes (i-iii) executed in 8 steps

(i) Inception definition (steps 1-3):

<u>Step 1: Define wastewater and solid waste selection criteria, system costs and the number of people per system</u>

System selection:

Within this framework the features of the residential area (population density and urban function) determine the solid waste and wastewater system (Table 5.2). In areas with high population densities and urban functions, like shopping malls, the environment receives higher pathogen and pollution loads (Mara et al., 2010). Consequently, suitable systems differ according to the type of residential area. This health and environment based system selection is not incorporated into the MDG. The MDG's only specify an access target, which is considered unsuitable given the above considerations (Baum et al. 2013).

Three types of wastewater systems are distinguished: on-site, community based systems (CBS) and off-site (medium-centralized or centralized) systems (Table 5.2). On-site systems typically serve one household, CBS typically serve 50-100 households, a medium-centralized off-site system serves up to 5,000 households, whereas a centralized system may serve up to 50,000 households. For more details reference is made to the Appendix Chapter 5, Section 1 and Kerstens et al. (2015).

Table 5.2 Basic system selection for wastewater treatment (WWT) and municipal solid waste (MSW) based on residential features. Typical households connected per system (hh/syst) are indicated in parenthesis. For MSW distinction is made between a conventional system (no resources recovery; only landfill) and a system with 3R (Reuse, Recycling and Recovery)

Residential features		Corresponding selected systems		
Status	Residential Population density (people pp/ha) [a]	WWT Systems (hh/syst) [b]	MSW systems [a]	
			Conventional: landfilling of waste	3R + landfill
Rural	Low (< 100)	On-site (1 hh/syst)	Not applied	Home compost
	High (> 100)	Community based (50-100 hh/syst)	Collection and landfill	Decentralized recovery with centralized landfill
Urban	Low (< 100)	On-site (1 hh/syst)		
	Medium (100-250)	Off-site: Medium-centralized (500-5,000 hh/sys)		Central recovery with landfill
	High (>250)	Off-site: Centralized (10,000-50,000 hh/syst)		

[a] indicated densities are general guidelines and can be adjusted (see also example Indonesia).

[b] For a description of WWT and MSW systems reference is made to Kerstens et al. (2015)

In this framework special attention is paid to poor communities. Based on population densities, urban slums could qualify for off-site systems. However, slums often lack water facilities, consist of temporary or non-legal houses, and residents are unable or unwilling to pay (Sijbesma, 2011). Therefore, off-site systems may not be a feasible option in the short term. Instead, temporary facilities such as community sanitation centers, where people may bath, wash and go to the toilet should be considered (Ulrich et al., 2009). These can be replaced by or adjusted to a more structural solution during planned renovation or rehabilitation (Bappenas, 2014a; USDP, 2015).

Municipal Solid Waste (MSW) system selection is based on residential features (Table 5.2) and distinguishes two types of system. In the first (conventional) system all collected waste is disposed of in a landfill. No interventions are planned for households in low density rural areas, since collection is not feasible due to access constraints and high collection travelling distances (MoPW - Ministry of Public Works, 2014a). In high population density rural and urban areas a conventional system is applicable. In the second (3R) system the amount of waste that requires landfilling is reduced by resource recovery in a so-called 3R program (*Reduction*, *Reuse*, *Recycling*) and may comprise plastic and paper recovery as well as compost and energy production from organic solid waste (Aprilia & Tezuka, 2010; Aprilia et al., 2011; Kerstens et al., 2015). Three scales of 3R are distinguished: home compost; decentralized (community level); and central (Table 5.2), involving composting, digestion and paper and plastic recovery. The recommended scale of 3R per type of residential area was based on an analysis that considered type and amount of waste, applicable technology, collection transfer and transport costs and treatment locations (Achillas et al., 2013; Kerstens et al., 2015) and is further elaborated in Appendix Chapter 5, Section 2.

Because of population dynamics, rural areas may become urban (BPS - Buro Pusat Statistik, 2010a), which affects wastewater or solid waste system selection. Therefore all current rural areas with a population density exceeding 25 pp/ha were regarded as future urban areas (Bappenas, 2014b; USDP, 2015). In this framework, priority can be given to sanitation development in urban areas over rural areas and larger cities over smaller cities in terms of targets and implementation of large infrastructures. For example, landfill development in metropolitan and larger cities may be prioritized over medium size and small cities, which is a consideration of the Indonesian government (Bappenas, 2014a).

Cost determination

To enable policy makers to budget and allocate funds, investment and operational costs for each defined WWT and MSW system are determined. This framework uses per capita investment and operational costs (Kerstens et al., 2015), which allows for a direct calculation of total costs based on the targeted population to be served (step 7 in Figure 5.1). Investments should include “hardware” (costs for installations, vehicles etc.), “software” (costs for studies, designs, socialization, health campaigning and advocacy), and land costs. Within a country, prices may vary depending on location, availability of materials and skills. Accordingly, these location dependent price levels should be included in the framework. Reference is made to Appendix Chapter 5, Section 3 and Kerstens et al. (2015). Costs for replacement of septic tanks and the provision of in-house piping were not included in the current analysis.

<u>Step 2: Define the planning term and nationwide sanitation targets</u>

The framework requires input on (1) future access, (2) urban system improvement, and (3) the planning horizon. The targets largely determine the number of facilities and corresponding budgets required (see also Step 7).

Access targets up until 2015 were typically based on the MDG’s (WHO & UNICEF, 2014). After 2015 governments should define their own targets (Bappenas, 2014a). In many developing countries, households apply poorly-performing septic tanks that are in fact leach-pits (WSP, 2013a) and that put public health at risk, especially when there is groundwater well use in the area (ADB, 2013a; Baum et al., 2013). Therefore, a switch from existing on-site systems to off-site systems is included when off-site systems are recommended for that area, following Table 5.2. The planner should define the percentage of people that will switch in discussion with the involved stakeholders. Since the associated health impact of providing access to people without a wastewater facility is larger than improving an existing, but poor performing facility (Hutton, 2013), priority is given to the former. The framework allows for a rapid identification of the impact of a certain switch factor or target in terms of required budgets, number of facilities and household connections, and thus supports an iterative calibration of this factor. At the same time, sustainable septic tank application requires also campaigning and advocacy and the development of sludge management facilities. Depending on the type of selected off-site technology, pathogen and pollutants removal, such as Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD), nitrogen (N) and phosphorus (P) may vary (Kerstens et al., 2015). Compared to septic tanks that only treat black water with poor removal efficiencies and cause diffused ground water pollution, off-site systems also treat greywater. As a tertiary step disinfection of treated effluent could be considered to further reduce health and environmental impact.

Finally, the planner should define a planning horizon, which often depends on national policies (GoI, 2006). Preferably a long-term horizon is selected since planned infrastructure is designed for long lifetimes and relates to long-term water policies (Ng et al., 2014). However, the longer the planning period, the more uncertain developments become (Ahn & Kang, 2014) and therefore short-term (typically 5 years) and medium-term (10-15 years) targets in the context of longer term ambitions (e.g. 20 years) should be considered.

Step 3: Define the roles and responsibilities per institution and system

The implementation of a WWT or MSW system requires a series of activities including feasibility studies, designs and construction for which different institutions are responsible. Identification of the responsible authorities will assist the planner in defining and allocating implementation budgets (Mara et al., 2010; Winters et al., 2014).

For each wastewater and solid waste system four features are determined:

(1) Costs of a specific system activity as a percentage of the total costs to implement that system;

(2) Origin of budget (National, Local, Private/users);

(3) Responsible department or (group) of ministries and;

(4) Activity cost type (hardware, software or land).

For example, the costs for a household connection may amount to (1) 15% of the total implementation costs of an off-site system. These costs are typically funded by (2) a local government, for example the Ministry of Public Works (3). The costs are regarded as hardware (4). To determine this cost allocation, an interactive process with decision makers from different departments (e.g. planning or public works) to produce a matrix as shown in Table 5.5 and Table 5.6 is suggested.

(ii) Data collection (Steps 4 and 5):

- Population data and forecasts can be retrieved from on-line databases or websites (UNpopulation, 2012; BPS, 2013, 2014; DSM - Department of Statistics in Malaysia, 2014; NBSC - National Bureau of Statistics in China, 2014; WHO & UNICEF, 2014);
- WWT and MSW management relates to human settlements and the framework uses residential densities (Table 5.2). Land use data can be estimated using available tools, such as "Google Maps". Land use data was also used to determine compost transport costs between urban and rural areas (see Appendix Chapter 5 Section 3);
- Current access data can be obtained through the Joint Monitoring Program (WHO & UNICEF, 2014), national databases (BPS, 2010b), health surveys (Ministry of Health, 2013) or directly from relevant ministries (Bappenas, 2014b; MoPW, 2014a).

(iii) Data Processing (Steps 6, 7 and 8):

In Step 6, the collected residential population data and urban/rural status (Step 4) for each defined area is used to select the system and its corresponding per capita costs, following Table 5.2. Selected systems per location are visualized in GIS.

In Step 7, the population that requires service provision in an area is the sum of (1) targeted future residents with access and (2) the residents living in high density urban areas that switch from an on-site system to an off-site system following Step 2 and corrected for the residents that already have access at the start (Step 4). The costs for implementation in one area are calculated using the specific per capita costs (Step 6) combined with the population that requires servicing.

In Step 8, the costs determined in Step 7 are allocated to responsible institutions using Step 3.

5.3 Methodology for data collection in Indonesia

Step 1. Define WWT and MSW selection criteria, costs and the number of people per system

WWT system selection followed the criteria defined in Table 5.2 with the additional criteria that new urban developments (population growth) in areas with population densities ranging from 25-100 pp/ha apply a medium-centralized WWTP, and that households in existing areas with densities of 25-100 pp/ha are served with septic tanks (Bappenas, 2014b). Population groups planned to apply on-site and CBS systems (Table 5.2) will be served by, respectively, septic tanks and anaerobic baffled + anaerobic filters technologies. Following discussions with the Indonesian Ministry of Planning, the applied distribution of technologies for medium centralized systems off-site systems was 40% anaerobic filters, 30% conventional activated sludge (CAS) systems and 30% CAS systems with enhanced N, P removal. For central systems the applied distribution was 10% anaerobic filters, 30% CAS systems, 30% CAS systems with enhanced N&P removal and 30% Aerobic Granular Sludge systems.

The number of households connected to a medium-centralized or centralized WWTP was determined based on the city size (Table 5.3) following discussion with the National Development Planning Agency.

Table 5.3 Households connected to a medium-centralized or centralized WWT system depending on size of the municipality (in 2035)

Off-site	Unit	Size of municipalities (households)			
		< 20,000	20,000-100,000	100,000-200,000	>200,000
Medium-centralized	Households connected	500	1,000	2,000	5,000
Centralized		10,000	10,000	25,000	50,000

In the MSW selection (Table 5.4) urban areas have priority and developments are planned to start in 2015. High density rural areas have no priority but will be served from 2020 onwards. Kerstens et al. (2015) describe centralized 3R systems applying only composting and digestion + composting to process the organic waste fraction. In this study half of the central 3R facilities apply digestion and composting and the other half only composting. Applied investments and net Operational Expenditures (OPEX) are indicated in the Appendix Chapter 5, Section 3 and were obtained from Kerstens et al., (2015). To determine the number of facilities required, decentralized and centralized 3R facilities were defined to serve, respectively 1,200 and 40,000 household (Bappenas, 2014b).

Table 5.4 MSW system selection for Indonesia as a function of density, urban/rural status and time of implementation

Residential features	Type of area	Rural		Urban			
	Population density (pp/ha)	<25	>25	<100	>100	<100	>100
Activity	Implementation period	Only after 2020		2015-2019		2020-2034	
	Collection	no	yes	yes		Yes	
	Level of 3R	Home	Decentralized	Decentralized	Central	Decentralized	Central
	Landfill (disposal)	no	yes	municipalities without an existing landfill [a]		Extend landfills	

[a] 3R related developments are started at the same time as landfill developments.

Construction prices differ per province in Indonesia and vary by a factors of 0.7 to 2.6 of the Jakarta price (TTPS - Tim Teknis Pembangunan Sanitasi, 2009). In the framework, provincial price corrections are included, but variations are reduced by 50%. This has been implemented because reported price differences were found to inaccurately reflect the cost for wastewater and solid waste facilities (MoPW, 2014a).

Step 2. Define planning boundaries and nationwide sanitation targets.

Urban and rural WWT and MSW access targets were defined for each 5 year period from 2015 until 2035. In this study two scenarios are compared that apply different targets for the first five years (2015-2019).

1. Scenario 1 follows the “universal access” targets.
2. Scenario 2 is a downscaled scenario 1, where only 75% of the targets defined in Scenario 1 are reached by the end of 2019, corresponding with 75% access to WWT facilities and 53% for urban MSW in 2019.

In both scenarios a 5% switch from current on-site system users to off-site systems in high density urban areas was defined until 2019 and a 50% switch by 2035 (Bappenas, 2014b; USDP, 2015). After 2020 both scenarios are the same and access is 100%, excluding the low density rural groups for MSW development.

Step 3. Definition of roles and responsibilities per institution and system.

Table 5.5 and Table 5.6 show the applied division of budget source for WWT and MSW facilities, differentiating (i) level of funding, (ii) department and (iii) type of activities. The use of this table is clarified by two examples (marked with the thick dashed line).

1. For on-site systems 15% of the total costs are related to general campaigning and advocacy. This budget is provided by the (i) national government (N), for (ii) campaign and advocacy (CA; referring to the Ministries of Health and Home affairs) and concerns (iii) software costs (S).
2. The percentage of house connection costs for medium-centralized systems was estimated as 13% of the total costs. This budget is provided by the (i) local government (L), of (ii) Public Works (PW) and concerns (iii) hardware costs (H).

Table 5.5 Applied Division (%) of budget [a] per (i) level of funding, (ii) department and (iii) activity for WWT facilities. The darkness of the colours shows the relative weight

Sub-sector	Wastewater															
System	On-site				CBS				Medium-centralized				Centralized			
Activity	%	source			%	source			%	source			%	source		
		i	ii	iii		i	ii	iii		i	ii	iii		i	ii	iii
Studies																
Master plan									0.25	N	PW	S	0.25	N	PW	S
Additional studies									0.25	Lo	PW	S	0.25	Lo	PW	S
Design																
guidelines					1	N	PW	S	1	N	PW	S	1	N	PW	S
detailing					4	U		S	3	N	PW	S	1	N	PW	S
Campaign, Advocacy																
General	15	N	CA	S	2	N	PW	S	1.5	N	CA	S	3	N	CA	S
Local	5	Lo	CA	S	4	Lo	CA	S	2	Lo	CA	S	1.5	Lo	CA	S
Land					11	U		La	3	Lo		La	3	Lo		La
Construction																
House connection	9	U		H	24	U		H	13	Lo	PW	H	11	Lo	PW	H
Sewer/ septage sludge processing facility [b]	1	N	PW	H	22	N	PW	H	43	N	PW	H	57	N	PW	H
Treatment	70	U		H	32	N	PW	H	33	N	PW	H	22	N	PW	H
All	100				100				100				100			

[a] Source codes i, ii and iii refer to:

i. Level of funding: national (**N**) or local (**Lo**) government or users/private (**U**);

ii. Ministry: Public Works (**PW**), Ministry of Health/Home Affairs (**CA**) for campaign & advocacy;

iii. Type of activity: hardware (**H**), software (**S**) or land acquisition (**La**)

[b] Sludge from on-site systems and community based systems is collected and processed in a central facility for which indicated costs are reserved. For septic tanks, designs guidelines are available (MoPW, 2000) and no additional studies and designs are required

Table 5.6 Applied Division (%) of budget [a] per (i) level of funding, (ii) department and (iii) activity for MSW facilities

Sub-sector	Municipal Solid Waste							
System	Collection- transport				Treatment			
Activity	%	source			%	source		
		i	ii	iii		i	ii	iii
Studies								
Master plan	3	N	PW	S	1.5	N	PW	S
Additional studies	1	Lo	PW	S	1	Lo	PW	S
Design								
guidelines	2	N	PW	S	2	N	PW	S
detailing	2	Lo	PW	S	3	Lo	PW	S
Campaign, Advocacy								
General	2	N	Ca	S	1.5	N	Ca	S
Local	4	Lo	Ca	S	2	Lo	Ca	S
Land	9	Lo		La	19	Lo		La
Construction								
Civil construction	52	Lo	PW	H	54	N	PW	H
vehicles	25	U		H				
treatment facilities					16	Lo	Pw	H
All	100				100			

[a] Source codes i, ii and iii refer to:

i. Level of funding: national (**N**) or local (**Lo**) government or users/private (**U**);

ii. Ministry: Public Works (**PW**), Ministry of Health/Home Affairs (**CA**) for campaign & advocacy;

iii. Type of activity: hardware (**H**), software (**S**) or land acquisition (**La**)

<u>Step 4. Data collection on population growth, land use, urbanization and poverty level</u>

Provincial population and urbanization projections for 2010-2035 (BPS, 2013) were applied to all (nearly 80,000) 2010 urban and rural administrative areas in their respective provinces (BPS, 2014). Residential land use and anticipated 2035 developments followed the Java Spatial Model (MoPW, 2011). For islands outside Java, the fraction of residential area was determined by extrapolation of the Java data, applying a correction factor of 0.75 to compensate for these less densely populated areas (see Appendix Chapter 5 Section 4). 2010 urban and rural poverty data (BPS, 2010b) were extrapolated to the 2015 population using the urban and rural projections (BPS, 2013), but excluding a possible decreasing poverty rate (Suryahadi et al., 2012). Maps with planned WWT and MSW system implementation were prepared in ArcGis10 (ESRI, 2010).

<u>Step 5. Collection of data on current access to and state of wastewater and solid waste facilities</u>

To determine the frequency distribution for rural, urban, poor and non-poor households with access to WWT facilities, the 2010 SUSENAS (National Socioeconomic Survey) data (BPS, 2010b) were aggregated in SPSS 17.0 using univariate data analysis. The overall access to WWT facilities at the end of 2014 (Table 5.7) was defined as 60% (BPS, 2013; Bappenas, 2014b; MoPW, 2014a; WHO & UNICEF, 2014). Provincial urban and rural MSW access data were collected from Ministry of Health (2010) and landfill and construction data from MoPW (2014). Data on existing sludge management facilities was obtained through the MoPW and the total number was corrected for malfunctioning plants (MoPW, 2012).

Steps 6-8 were then processed as explained in Section 2, (iii) Data Processing.

Table 5.7 Indonesian WWT access data for Urban, rural, poor and non-poor households 2010 (SUSENAS) and 2014 (defined in this study) [a]

Fraction of households with access	Poor total		Non-poor total		Urban total		Rural Total		Nationwide	
	2010	2014	2010	2014	2010	2014	2010	2014	2010	2014
Open defecation	35%	32%	17%	15%	9%	8%	31%	29%	20%	18%
Unimproved	29%	26%	23%	21%	18%	16%	30%	28%	24%	22%
Improved	36%	42%	61%	64%	73%	76%	39%	42%	56%	60%

Fraction of households with access	Urban		Rural		Urban		Rural	
	poor				non-poor			
	2010	2014	2010	2014	2010	2014	2010	2014
Open defecation	23%	20%	42%	39%	7%	6%	27%	26%
Unimproved	22%	20%	32%	29%	17%	16%	30%	28%
Improved	55%	60%	26%	32%	75%	78%	43%	46%

[a] BPS (2010b) identifies: (1) open defecation (no latrine), (2) unimproved access (latrine, but no gooseneck or treatment) and (3) improved access (combination of latrine, gooseneck and treatment)

5.4 Results

5.4.1 Selected systems per location in GIS

Off-site wastewater (Figure 5.2) and centralized 3R solid waste systems (Figure 5.3) are found around the bigger cities, like Jakarta, Surabaya, Bandung, Yogyakarta and Surakarta. In the yellow marked medium density urban areas (25-100 pp/ha), the current population will be served by on-site systems, but new developments by a medium-centralized system. Outside these urban areas on-site (light green), Community Based Systems (CBS) (dark green) wastewater systems and home composting (green) and decentralised 3R (light brown) solid waste systems are planned. Some areas in Indonesia are categorized as “forest /national parks/lakes” (no registered inhabitants) for which on-site systems are assumed for wastewater, whereas for solid waste these areas are indicated separately.

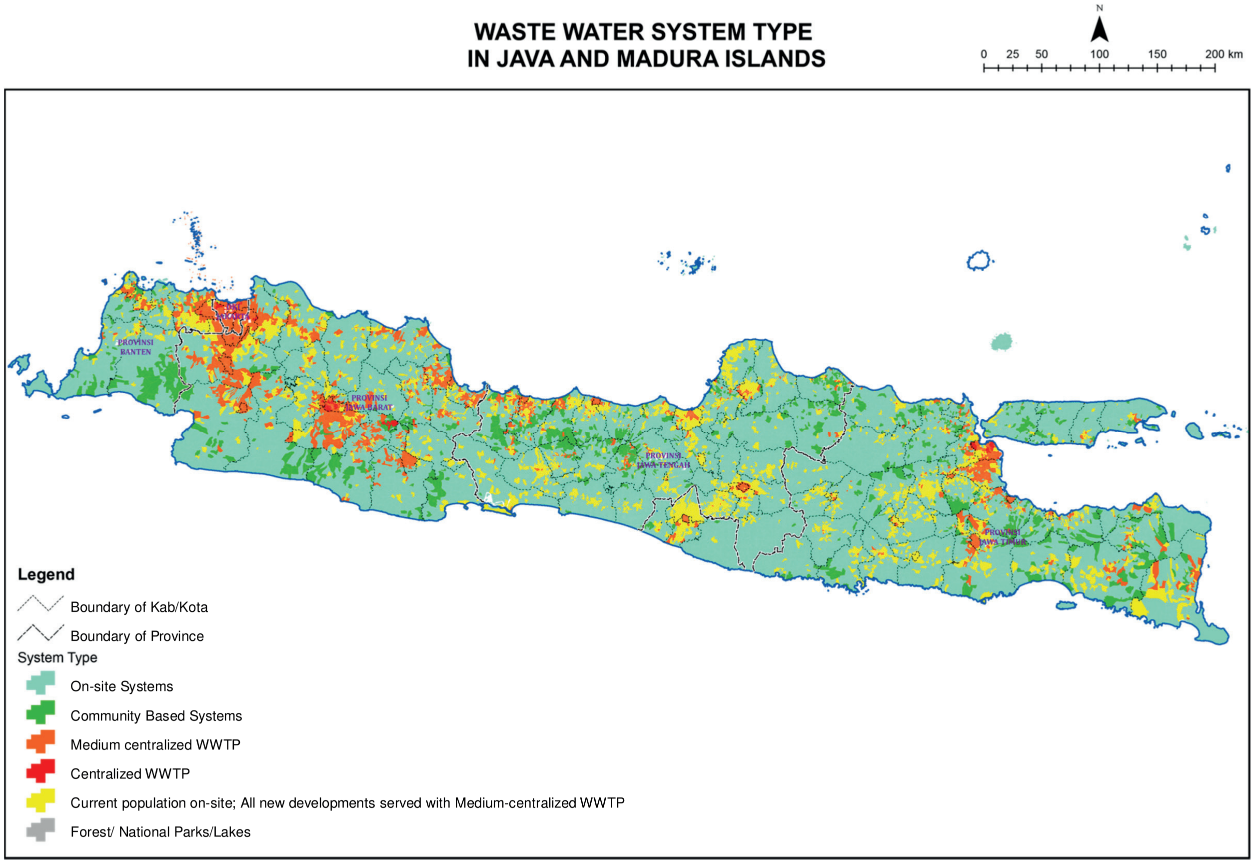

Figure 5.2 Java planned WWT distribution (2025). Light green is on-site; dark green is Community Based; orange is medium-centralized WWTP; red is centralized WWTP; yellow is on-site for current population and medium-centralized WWTP for new development; Forest/national parks/ lakes apply on-site systems

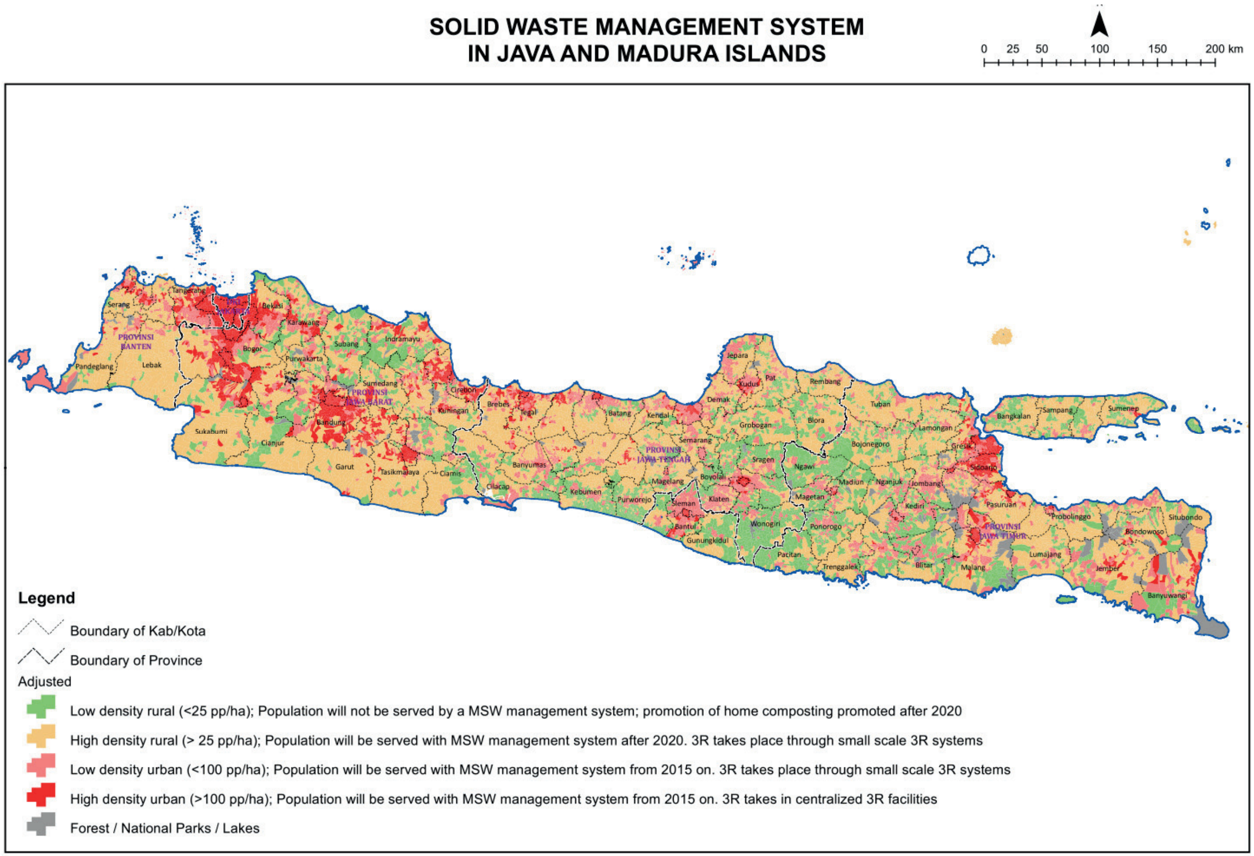

Figure 5.3 Java planned MSW system distribution in 2025; green is home composting; light brown (High density rural areas) and pink (low density urban areas) show decentral 3R stations; Red is high density urban area for central 3R facilities; Forest national parks and lakes (grey).

5.4.2 Numbers of people served and costs

Planned development of access to WWT and MSW systems

It is planned that by 2019 the majority of the population (230 million; 85%) is served by an on-site system, nearly 5 million people (2%) by CBS, 31 million people (12%) by a medium-centralized and 4.5 million (2%) by a centralized WWTP in Scenario 1 (Figure 5.4 A). In Scenario 2, planning shows 180 million people connected to an on-site system, 3 million to CBS, 19 million to a medium-centralized and 2.5 million to a centralized system, whilst 67 million (25%) remain without access (Figure 5.4 B). By 2035, planning shows over a third of the population would be connected to a medium-centralized or centralized system (Figure 5.4 A and B and Appendix Chapter 5, Section 5).

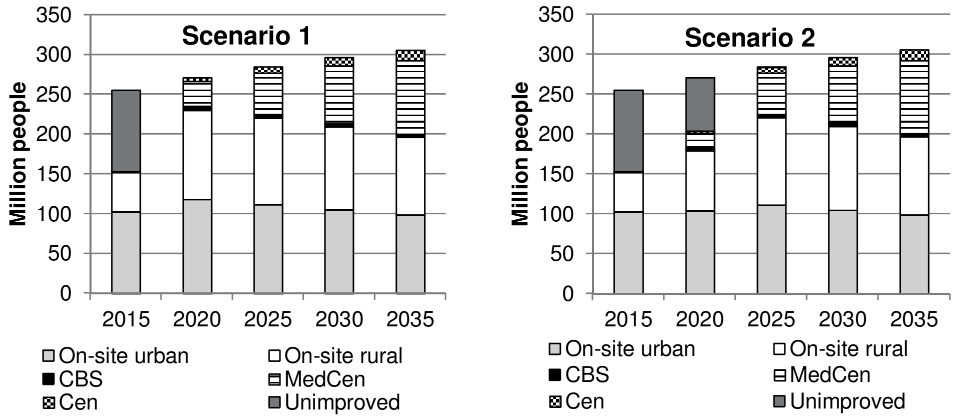

Figure 5.4 A (Scenario 1; left) and **B** (Scenario 2; right) WWT access development with urban and rural on-site systems, CBS and MedCen (medium-centralized) and Cen (centralized) systems. In Section 5 of the Appendix Chapter 5 numeric values are presented

Access to solid waste collection and treatment (including disposal) services for the two scenarios (Figure 5.5 A and B) show that rural collection is not planned to commence until 2020. By 2020 collection and treatment is planned to serve 113 million people in Scenario 1 (Figure 5.5 A), and 87 million people in Scenario 2 (Figure 5.5 B). After 2020 a higher rural treatment rate than collection is achieved as no collection system is planned for low density rural areas, but some households apply home composting to reduce 20% of landfilled waste.

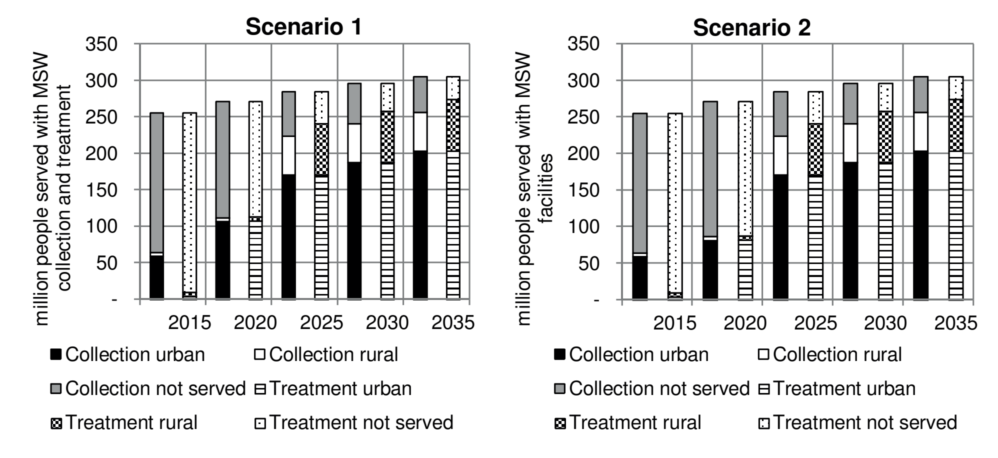

Figure 5.5 A (Scenario 1; left) and **B** (Scenario 2; right) MSW access development for collection and treatment services. In Section 5 of the Appendix Chapter 5 numeric values are presented

Until 2019, 2,360 and 1,590 medium-centralized systems are required in the two scenarios (Table 5.8), whereas this number will increase to a planned 6,720 systems by 2034. The difference in centralized systems after five years is smaller (53 versus 48), because construction of centralized systems will start even if not all people are connected.

In Scenario 1 (benefiting 103 million urban people, Figure 5.5 A) a total of 236 landfills (Table 5.8) are planned of which 105 are new while 131 already exist (MoPW, 2014b). In Scenario 2 an additional 45 landfills are needed to serve 87 million people. This corresponds with an average of about 1 million (Scenario 1) and nearly 2 million persons per landfill (Scenario 2).

Table 5.8 Development of the cumulative number of WWT and MSW systems for Scenarios 1 and 2 in 2019

Scenario	year	WWT systems					MSW		
		On-site	CBS	IPLT [a]	MedCen	Cen	Decentral 3R	Central 3R	landfill
Scenario 1	2019	46,950,000	13,300	750	2,360	53	2,460	165	236
Scenario 2		36,810,000	7,900	630	1,590	48	1,850	146	176
100% access	2034	48,870,000	13,500	780	6,720	91	7,690	630	490

[a] IPLT = (central) sludge management facility for treatment of on-site and CBS sludge. Presented number included the existing 88 identified functioning IPLT (MoPW, 2012)

Planned WWT and MSW budget requirements

Estimated investments for the 2019 Scenario 1 and 2 scenario are 25 and 13 billion US$ (Figure 5.6) respectively. A total of 56 billion US$ is required until 2034 (Figure 5.6). For Scenario 1, nearly 20 billion US$ or 79% of all investments (2015-2019) are required for WWT facilities and nearly half of that amount (8.7 billion US$) is for medium-centralized WWTP investments. Nearly

80% of the investment in off-site systems relates to implementation of sewer systems (see Appendix Chapter 5, Section 3). Although in this study standardized per capita unit prices were used (Kerstens et al., 2015), sewer system prices depend on population densities, type of sewer systems, their materials and topography and may therefore differ in practice and should be confirmed in a design and costing phase (Loetscher & Keller, 2002; Petit-Boix et al., 2014). The technical and institutional operation and maintenance (O&M) budget requirements for solid waste is nearly 1.5 times higher than that for wastewater systems by 2019 and 1.8 times by 2034 (Figure 5.6 A and B). This higher solid waste O&M budget is the result of the solid waste collection costs that may contribute up to 70% of the operational costs (Kerstens et al., 2015).

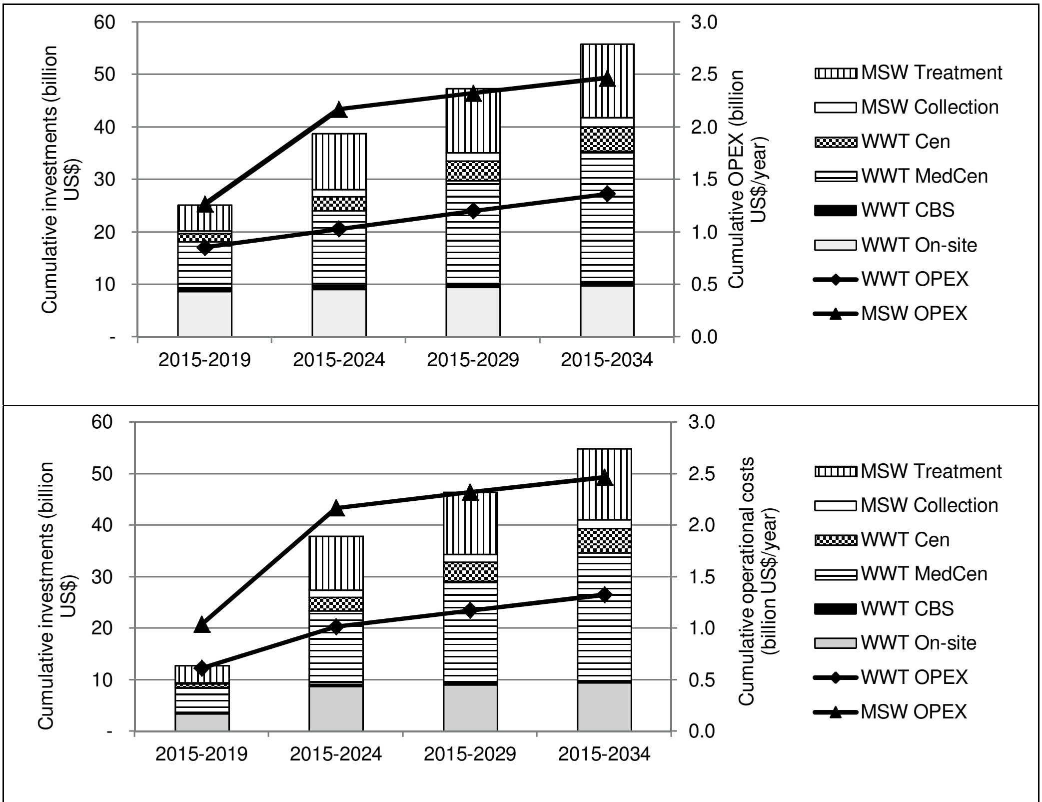

Figure 5.6 A (scenario 1; above) and **B** (scenario 2; below) WWT and MSW cumulative investments (primary Y-axis) and operational costs (OPEX) at the end of each indicated period (secondary Y-axis)

WWT budget requirements for poor and non-poor households

The calculated WWT budgets benefiting poor people in Scenario 1 and 2 are 4.4 and 2.0 billion US$ respectively in the period 2015-2019 (Figure 5.7). This corresponds to about 23% of the total WWT facilities investments, whereas 18% of the Indonesian population was poor (BPS, 2010b).

The WWT investment related to rural poor households was 29% of the total rural WWT investment, whereas 26% of the rural population was poor.

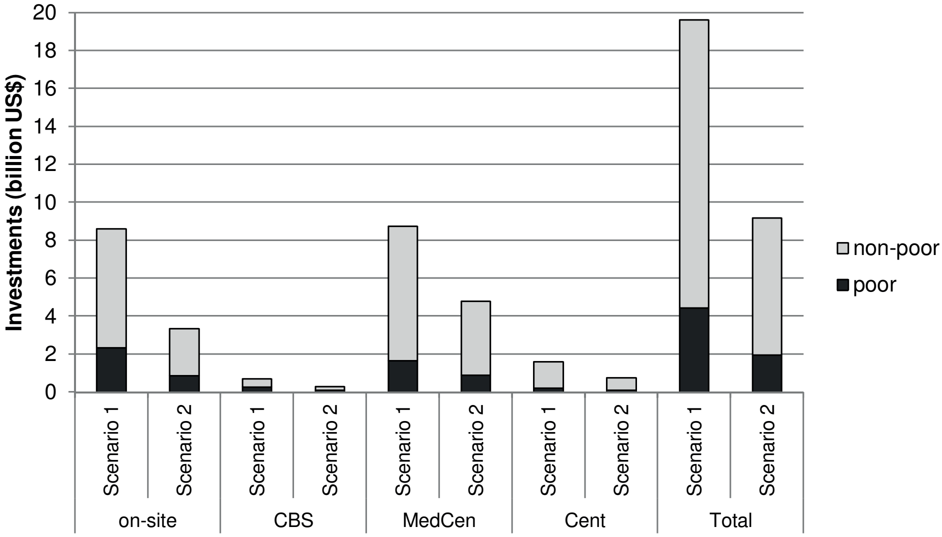

Figure 5.7 Calculated investments (2015-2019) benefiting poor and non-poor households per type of WWT system for Scenario 1 and 2

5.4.3 Budget breakdown

For Scenario 1 and 2 a contribution of 53% and 57%, respectively, from national government and 18% and 21% from the local Government was determined (Table 5.9) using the division formulated in Table 5.5 and Table 5.6. The remaining part of the budget is expected to be paid by users directly and predominantly comprises investments for septic tanks.

Table 5.9 2015-2019 Scenario 1 and 2 budget division per level (national, local or users) and department and activity (PW-Public Works for infrastructure and design/ studies; MoH/HA Ministry of health and home affairs for Advocacy and campaigning) for Indonesia

Level [a]	Origin of funds (billion US$)	Scenario 1				Scenario 2			
		WWT	MSW	Total	Fraction	WWT	MSW	Total	Fraction
Total	National	10.2	3.1	13.3	53%	5.3	2.0	7.3	57%
	Local	2.3	2.2	4.6	18%	1.2	1.4	2.6	21%
	users	7.2	0.1	7.3	29%	2.8	0.1	2.8	22%
	total	**19.8**	**5.4**	**25.2**	**100%**	**9.3**	**3.5**	**12.8**	**100%**
National	Infra (PW)	8.3	2.8	11.1	84%	4.4	1.8	6.3	86%
	Design/studies (PW)	0.4	0.2	0.6	5%	0.2	0.1	0.4	5%
	Adv. Ca. (MoH/HA)	1.5	0.1	1.6	12%	0.6	0.1	0.7	9%
	Total	**10.2**	**3.1**	**13.3**	**100%**	**5.3**	**2.0**	**7.3**	**100%**
Local	Infra (PW)	1.7	0.5	2.1	46%	0.9	0.3	1.2	44%
	Design/studies (PW)	0.0	1.7	1.7	37%	0.0	1.1	1.1	42%
	Adv. Ca (MoH/HA)	0.7	0.1	0.8	17%	0.3	0.1	0.4	14%
	total	**2.3**	**2.2**	**4.6**	**100%**	**1.2**	**1.4**	**2.6**	**100%**

[a] The sum of the National and Local budgets in the first level (Total) are split up in the second level (National) and third level (Local)

5.5 Discussion

5.5.1 Advantages and novelty of the proposed framework

The lack of an existing functional planning tool resulted in the development of the presented framework that enables nationwide planning in the wastewater and solid waste sector. The output directly links national policies with geographical information, number of facilities and budget required per responsible implementing institution. In contrast to most existing frameworks, the presented framework is inclusive of all population groups (urban, rural, poor and non-poor) in a country. It further, provides politicians and planners with meaningful insight into the viability of political ambitions and allows these decision-makers to anticipate the numerous consequences of their choices which are elaborated in the following paragraphs.

First, the GIS outputs (Figure 5.2 and Figure 5.3) show where centralized wastewater and solid waste facilities coincide and allows for combined treatment of wastewater and solid waste flows in one central facility. This may have favourable consequences in terms of energy requirements, removal efficiencies and quality of compost produced (Koné et al., 2007; Zitomer et al., 2008; Kerstens et al., 2015). Other benefits of a combined infrastructure development can be found in joint management and sharing of facilities (offices, access roads), as was shown in Banda Aceh, where organic solid waste digestion, composting and landfilling are developed at the site of the septage sludge processing facility (USDP, 2015). In addition, locations were identified where construction of sewer systems can be integrated with the overall development of the area resulting in costs saving.

Second, the impact of policy choices on applied urban and rural systems was demonstrated. The result of prioritizing access to wastewater facilities before improving existing systems is an initial increase in the number of households having access to on-site systems. After 2020 the number of on-site systems decreases, as people living in high density urban areas with on-site access switched to off-site systems. With time progressing, the number of people planned to be connected to a centralized or medium-centralized system increased, because of (1) urban population growth, and (2) the switch from urban on-site users to off-site users. Comparing Scenario 1 and 2 shows that lower targets correspond with lower required budgets and vice versa. For example, with lower budget provision for rural sanitation development, less people or villages can be reached.

Third, the impact of prioritizing development of landfills on city size, serving a small fraction of the population (Scenario 2) results in an almost double total landfill capacity when compared to Scenario 1. The planned number of facilities (Table 5.8) and their location also allows the planner to anticipate the large scale implementations and land requirements. Land availability has been identified as a barrier in the development of large infrastructure as well as smaller infrastructure in urban areas (Aprilia et al., 2012; ADB, 2013a), and space saving has been a driver for the MSW 3R program (MoPW, 2014a). Similar large developments of medium-centralized systems took place in Malaysia, where between 2000 and 2008 about 300 WWTPs were built yearly (Haniffa et al., 2009). This large implementation of medium-centralized WTTP in Asian urban areas has been attributed to (1) potential lower costs than centralized systems, (2) possibilities for phased development and (3) potential for resource recovery (Starkl et al., 2012; Lapid, 2012). Malaysian developments showed the need for a nationwide investment strategy, reservation of land, standardization of sewer and wastewater facilities and structured training & certification programs (Haniffa et al., 2009).

Fourth, the presented framework enables the policy maker to determine required investments for poor households and to identify additional support needs. The rural poor have least access to wastewater facilities (Table 5.7). Large variation in access between poor and non-poor populations in Indonesia were further demonstrated by WSP (2014). Rural sanitation development in Indonesia follows the community-led total sanitation (CLTS) approach (Mehta & Movik, 2010; Mara et al., 2010). Since the approach largely relies on private (household) investments, allocation of an additional portion of the national budget to support rural wastewater access developments should be considered, especially given that this will benefit the most disadvantaged part of the population. Such a government supply driven approach combined with a demand driven CLTS approach has been successfully applied in Lao (ODI, 2011).

Fifth, the output of this framework provides guidance on required budget increases per level (national or local) of government and department. The 2014 national government sanitation budget in Indonesia was 1.5 billion US$ (USDP, 2014). The planned national and local budgets (Table 5.9) for Scenario 1 requires nearly 18 billion US$ over five years or 3.6 billion US$/year and corresponds with a factor 2.4 increase. The local government budget in 2014 was nearly 0.9 billion US$ (USDP, 2014) and for 2015-2019 remains at the same level, whereas the national budget requires a much higher increase from about 0.5 to 2.7 billion US$ per year (factor 5.4 increase). The required implementation and operation sanitation budget may be compared with the socio-economic impacts that will accrue to society, such as improved health conditions, reduced travel and waiting time for personal hygiene and increased property values. If these benefits exceed the costs of implementation, this can feed into advocacy efforts to further raise funding from governments, households and the private sector (Hutton, 2013).

Sixth, the framework helps to identify the budget to create an "enabling environment" to accelerate access to wastewater and solid waste facilities. Studies have recommended strengthening (1) policy frameworks and enforcements, (2) institutional arrangements and capacities, (3) creating demand and accountability and (4) promotion of public debate and communication (ADB, 2013a; Kearton et al., 2013; WSP, 2013b, 2014; USDP, 2014; Winters et al., 2014). Thus far, off-site systems in Indonesia have been implemented with limited success, especially with respect to connecting households (USAID 2006; Whittington et al. 2000; Kearton et al. 2013). By subsidizing household connections (see Table 5.5) an increase in connections may be facilitated. The required (national + local) budget reservations for campaigning and advocacy, amounted to 2.4 billion US$ for five years (Table 5.9) corresponding to 10% of the total sanitation budget. This is 5 times the current communication health budget (Bappenas, 2014b). A current challenge in Indonesia is underperforming septage sludge management, which has been attributed to the lack of demand for services from households, poor technical and financial management and a lack of dissemination capacity within the national government (WSP, 2013b). The latter was addressed in the current framework by including investments and operational costs of septage management, including collection, as well as advocacy and campaigning costs.

Finally, the framework supports policy makers in the determination of tariffs for wastewater and solid waste services. Operation and maintenance budgets are now paid by local governments, but eventually should be paid by users themselves through fees. The willingness to pay for solid waste services has been established in Indonesia – more so than for wastewater services (WSP, 2011). However, resistance to pay for both services can still be found (Winters et al., 2014) and underlines the need for campaigning and advocacy.

5.5.2 Linking the framework to current sanitation practice

Besides the absence of a functional sanitation planning tool, other barriers in sanitation development in developing countries are: inadequate sanitation regulatory frameworks and cross-sector policy coordination, rapid urbanization, low community awareness on the importance of sanitation, land availability, limited local capacity to assure operation of facilities, and inadequate investments in sanitation systems (ADB, 2013a; Kearton et al., 2013).

Except for regulatory barriers, the presented framework may strengthen the above listed shortcomings in terms of planning investment, division of responsibility and budget reservations for operation and maintenance. However, in the elaboration of a detailed plan (e.g. at city or district level), other factors should be considered such as capacity of local institutions, land availability, especially for landfill developments, topography, ground water levels, the skill level of operators, specific demands by a municipality, requirements from any foreign donors and desired level of household participation (Loetscher & Keller, 2002; ADB, 2013b).

Because the framework was developed using information from the smallest available administrative units, the presented nationwide results can also be broken down to lower levels of governments, such as cities and provinces. In Section 6 of the Appendix Chapter 5, three examples of a "translation" of national planning to local levels are presented for WWT and MSW budgets, systems and percentage of investments contributing to poor households for the period 2015-2019. In Indonesia, cities must prepare a City Sanitation Strategy to be eligible for national funding. Thus, a 5-year plan, using a similar residential area-based sanitation system selection as presented in the framework, is prepared to formulate budgets and specify required institutional and advocacy and campaigning activities (Kearton et al., 2013; Parkinson et al., 2014; USDP, 2014). An example of such a CSS for the city of Tegal in Central Java is presented in Appendix Chapter 5, Section 7. This simultaneous development of a top-down supply and bottom-up demand for sanitation funding links the Indonesian central government's policy making and oversight role with the local governments' role for implementation (ADB, 2013a).

The framework developed in this study can be applied in other developing countries ,because most of the required input data is readily available through on-line databases (DSM, 2014; NBSC, 2014), and UN (United Nations) reports (UNpopulation, 2012; WHO & UNICEF, 2014).

5.5.3 Outlook to further developments

In presented application of this framework we focused on technologies that fit the current Indonesian context. However, in the future or for countries at a different development stage, other technologies may be feasible and their implementation can be accommodated in the framework provided their feasibility is linked to residential features and the investment and operational costs. Thus, the application of more advanced household level or community level WWT systems, including (1) nutrient removal as applied in South America or Japan (Tsuzuki, 2006; Aiyuk et al., 2006), (2) separate collection and treatment of grey water (Chen & Wang, 2009; Kerstens et al., 2009) or (3) replacement of conventional septic tanks by better performing UASB septic tanks (Kujawa-Roeleveld et al., 2005) can be included.

New sanitation systems that aim to recover resources following source separation (Larsen et al., 2009; Zeeman & Kujawa-Roeleveld, 2011; Tervahauta et al., 2013) were excluded in this study, as required separation of wastewater streams at household level and corresponding dual transport lines were considered too "high tech" for Indonesia, but can be fitted into this framework as well. Spiller et al. (2015) proposes implementation of technologies and infrastructure which are flexible, adaptive and robust in order to ensure the sustainability of these systems under dynamic conditions. These may comprise modular or prefabricated systems to cope with changing capacities or an addition of a post treatment step for nutrient removal or disinfection to meet future effluent requirements. Alternatively these may involve the use of utility tunnels in high density urban areas that allow for sewer system expansions or modification in a later stage. Integration of these flexible technologies in the framework should be further explored.

To link planning to a sanitation impact (e.g. improved surface water quality), the framework and its outputs may be combined with pollutant removal efficiencies, resource recovery potential and costs of WWT or MSW systems (Kerstens et al., 2015). As an example, cumulative discharged BOD, TN and TP loads on a national scale for Indonesia (2020) and on provincial scale for Jakarta (2035), using on-site systems, CBS and different four off-site technologies are compared as possible technical interventions. Figure 5.8 shows that on a national scale the impact on reduced discharge of BOD, N & P of a nutrient removing system like a Conventional Activated Sludge with and without enhanced N&P removal (CAS and CAS_N&P) or the Aerobic Granular Sludge (AGS) compared to a low cost anaerobic filter (An. Fil.) is limited (approximately 10% maximum) for the 2020 planned implementation. This is because only less than 15% (see Figure 5.4 A) of the population in 2020 is planned to be connected to an off-site system and the remaining part to septic tanks with limited N&P removal. This may suggest that the incremental improvement to the environment does not outweigh the higher cost of nutrient removing systems. However, in a heavily urbanized area where by 2035 60% of the population is planned to be connected to an off-site system, such as Jakarta province, the discharged N&P is reduced by over 40% using nutrient removing systems. This insight can affect the choice for a WWT system to apply in heavily urbanized areas. The framework thus fits in the need for a planning and design paradigm that can resolve trade-offs across spatial scales, temporal scales, and sustainability dimensions (Guest et al., 2009).

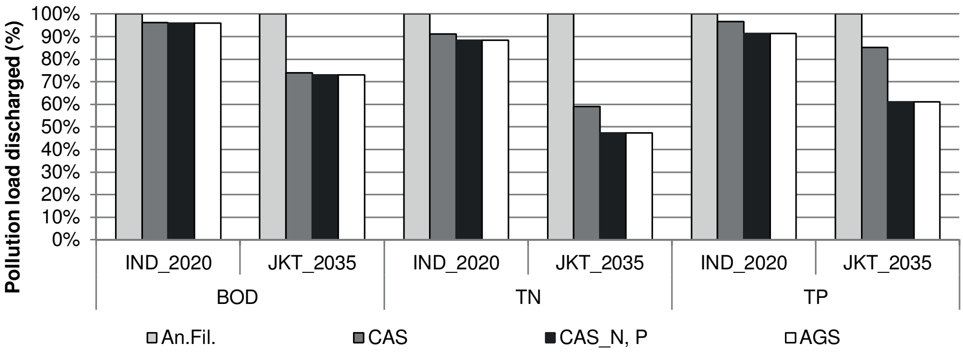

Figure 5.8 Comparison between on-site, CBS and indicated off-site technologies (Anaerobic filters, Conventional Activated Sludge (CAS), CAS with enhanced N, P removal (CAS_N,P) and Aerobic Granular Sludge (AGS)) on discharged pollutants (BOD, TN, TP) for Indonesia (IND) in 2020 and Jakarta Province (JKT) in 2035 following Scenario 1

5.6 Conclusions

A nationwide wastewater and solid waste planning framework was developed that links national policies to the required budget and facilities for different groups of beneficiaries (urban/rural and poor/non-poor) and for different geographical locations, whilst only using readily available and retrievable data.

Planned infrastructure was visualized in GIS and supports planners in prioritizing regional implementation. It further allows for identification of potential cost effective planning and implementation by combination of wastewater and solid waste treatment facilities or integration of sewer developments in anticipated residential area developments.

Application of the tool showed how policy choices (targets, prioritization) impact the number of facilities and budgets required. These insights can help in the formulation of additional programs to provide additional attention to poor households.

The framework was applied for Indonesia and outputs were adopted in the “National Medium Term Development Plan”. Required budget to reach universal access by 2019 was assessed as 25 billion US$ in 5 years. The contribution of the national budget required a more than fivefold increase compared to current budgets. The budget for campaigning and advocacy to strengthen the enabling environment was determined to be 10% of the total budget.

The framework can be used and further extended to resolve trade-offs across spatial scales, temporal scales, and sustainability dimensions.

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Appendix Chapter 5

Section 1 Description of wastewater systems and selection criteria

System description

In paragraph 5.2 (Table 5.2) of Chapter 5 reference was made to three types of wastewater systems that are distinguished in the framework. These will now be further elaborated:

- *On-site systems* do not require a sewer system and typically serve one household only. In this framework the applied on-site system is a septic tank, which is also the minimum treatment requirement In Indonesia (BPS, 2014);
- A *Community based system* (CBS) serves typically 50-100 households and in this study an average of 80 households was assumed. Wastewater is collected using a simplified sewer system (Mara & Broome, 2008). Treatment typically occurs in an anaerobic baffled reactor + anaerobic filter (Ulrich et al., 2009);
- A medium-centralized *off-site system* serves up to 5,000 households, whereas a centralized system may serve up to 50,000 households in which wastewater is collected using a simplified or (pumped) piped-collection system (Kerstens et al., 2015). The number of households connected to a medium-centralized or centralized WWTP was determined based on the city size (Table 5.3 of Chapter 5).

The described systems and their treatment and financial performances in the Indonesian context were analyzed by Kerstens et al. (2015). Septic tanks have the lowest total lifecycle costs (TLC), but show only a limited pathogen removal (~1 log), and low COD, BOD, nitrogen (N) and phosphorus (P) removal. The CBS systems have higher COD, BOD and pathogen removal than the septic tank, but limited N, P removal and their TLC are about 1.7 times higher than septic tanks (Kerstens et al., 2015). Applicable treatment technologies for off-site systems may range from anaerobic filters (similar performance as CBS) to membrane bioreactors (high removal efficiencies). Off-site systems have TLC of averagely 2.4 and 3.9 times the TLC of CBS and septic tanks, respectively (Kerstens et al., 2015).

System selection

Evidence suggests that there is a causal link between population density and urban functions in the vicinity of e.g. cinemas, shopping malls or Commercial Business Districts and the occurrence of diarrhoea (Lasut et al., 2008; Gondhalekar et al., 2013). Consequently, the system of choice differs per type of residential area. In this framework, <u>off-site</u> options are only considered in more densely populated areas, when on-site treatment could entail direct risks to public health, or when the risk exists of fecal contamination or eutrophication (UNEP (United Nations Environment Programme), 2004). The selection of type of off-site treatment technologies depends, among others, on applicable effluent regulations and land availability and is usually determined in a location specific feasibility study (MoPW, 2013a).

In rural areas or lower density urban areas (<100 people/ha) the minimum standard (on-site systems) with the lowest costs is recommended. In areas with increasing population densities, like peri-urban areas or higher density rural areas, CBS are proposed, because these show better performance than on-site systems but still have considerable lower costs than off-site systems (Kerstens et al., 2015). Reference is made to Table 2 of Chapter 5 for criteria.

Both septic tanks and CBS require regular emptying after which the sludge is treated in a central sludge processing facility. Consequently, in the planning of septic tanks and CBS the development of sludge processing facilities should be included as well. However, sludge collection in remote rural areas (for Indonesia < 25 pp/ha was applied) may not be feasible due to long transportation routes (Bappenas, 2014) and no facilities are planned for these areas.

Section 2 Description of Municipal Solid Waste (MSW) systems

Municipal Solid Waste (MSW) system selection is based on residential features (Table 5.2 of Chapter 5) and distinguishes conventional system and systems applying *Reduction*, *Reuse*, *Recycling* (3R) of waste. In low density rural areas only promotion of home composting is considered (Mehta & Movik, 2010), whereas for higher density populated rural and urban areas also digestion of organic waste and recovery of plastic and paper is considered.

In high density areas, a central 3R facility (outside the city) is more financially attractive than (multiple) decentral stations (Kerstens et al., 2015). Large area requirements of decentral 3R stations in land-scarce high density urban areas result in high investments and a consequential barrier in the development of these systems (Aprilia et al., 2012; Kerstens et al., 2015). In addition, in high density urban areas there is less direct demand for recoverable products (e.g. compost), because of the absence of agricultural activities (BPS, 2010). In medium density areas with lower land costs and a potential direct sale of recovered resources to nearby farmers, decentral 3R facilities are proposed (Kerstens et al., 2015). Compost is especially demanded by the horticultural sector that is often located nearby urban areas (Aye & Widjaya, 2006; Indraprahasta, 2013).

The division of households that apply a conventional or a 3R-based MSW system may have been formulated by the government (Bappenas, 2011) or, alternatively, should to be defined by the planner. In case a 30% target is defined and a low density urban area is considered, the framework plans that 30% of the population is served by a decentralized 3R station and 70% by a conventional system.

Section 3 Basis for wastewater and solid waste costing

In paragraph 5.2 of Chapter 5 the need for cost determination was introduced. Table A5.1 and Table A5.2 show the applied investment and operational costs of WWT and MSW facilities. Presented prices in Table A5.1 and Table A5.2 reflect Jakarta prices. For each province these were adjusted using a price correction in which variation compared to the Jakarta prices were reduced by 50% (TTPS, 2009).

The investment costs include “hardware” (costs for installations, vehicles etc.), “software” (costs for studies, designs, socialization, health campaigning and advocacy) and land costs. Operational costs comprise technical (e.g. energy, sludge, chemicals, maintenance) and institutional costs (management of facilities, operators and administrative tasks). In case resource recovery is applied, benefits from sale of recovered products are deducted from the operational costs following the values and assumptions given by Kerstens et al. (2015).

Costs for septic tanks (Table A5.1) include costs for sludge management facilities that were based on the guidelines of the Ministry of Public Works and include collection vehicle costs (MoPW, 2013a). Prices have been further checked with representatives of IUWASH (Indonesia Urban Water, Sanitation and Hygiene Project) and WSP (Water and Sanitation Program of the World Bank) that studied septage management in the Indonesian context in depth (WSP, 2013). Off-site system costs were based on a system distribution discussed with Bappenas (2014b) and Ministry of Public Works (MoPW) (2014a) . A sludge processing facility serves a maximum of 200,000 households. In case more households need to be served, multiple facilities are planned to be constructed.

Off-site system investment costs consider sewer costs, land costs and treatment costs. In the cost definition of off-site sewer systems the planner should make a distinction between existing and new “green field” residential area developments. In a green field situation, sewer systems can be integrated with other developments, like electricity and water supply and road and pedestrian path development and therefore require only about half of the costs compared to an existing area (Rioned, 2007). Hence, a 50% cost reduction for off-site systems sewer systems in new urban development compared to existing urban areas is applied.

Table A5.1 Applied investment and operational expenditures (OPEX) per on-site, CBS, medium-centralized and centralized WWT system for existing residential areas and new area to be developed in the future [a, b]

Parameter		Unit	on-site	CBS [c]	Medium-centralized [d]		Centralized [e]	
					Existing area	New development	Existing area	New development
Investments	Sewer	US$/cap	0	114	229	115	324	162
	WWTP		100	74	113	113	92	92
	Total		100	188	342	228	416	254
OPEX		US$/cap/year	2.4	3.0	8.0	8.0	6.7	6.7

[a] Applied investment and operational costs are rounded numbers and based on Kerstens et al. (2015); [b] Residential features refer to the projected 2025 (mid- term) data of each area; [c] The applied CBS sewer system is a simple and shallow one (Mara & Broome, 2008) and no price difference between existing and new development areas was applied; [d] Applied distribution is 40% anaerobic filters, 30% conventional activated sludge (CAS), 30% CAS with enhanced N, P removal; [e] 10% Anaerobic filter, 30% CAS, 30% CAS N&P, 30% Aerobic Granular Sludge (Kerstens et al., 2015)

Applied investments & net Operational Expenditures (OPEX) for MSW systems are shown in Table A5.2.

Table A5.2 Applied investments & net Operational Expenditures (OPEX) for MSW systems [a,b]

Parameter	Unit	Conventional (only landfill)		3R home	3R decentral		Centralized 3R	
		Rural	Urban	Rural[c]	Rural	Urban	Compost	Digest & Compost
Investments	US$/cap	45.2	72.9	2.6	73.3	95.3	57.0	69.8
Net OPEX	US$/cap/year	8.7	10.2	-1.3	4.6	5.1	5.9	5.2

[a] Applied investment and (net) OPEX are based on Kerstens et al. (2015).

[b] Transport of compost from central 3R facilities to rural areas is calculated separately based on average distances (MoPW, 2011) and transport costs of 0.70 US$/km after correction for increased fuel prices and wages since of the 2009 (0.40 US$/km) costs (Suletra et al., 2009);

[c] the price of a home composter (serving 5 people) was 13 US$ (MoPW, 2013b)

Transport costs

Transport costs of recovered resources products (compost) between urban and rural areas were determined in a model using land use data (Figure A5.1). The model assumed a centric municipality with the urban residential areas in the centre surrounded by the non-residential urban areas. These latter areas can potentially be used as locations for urban agriculture development (Indraprahasta, 2013). A rural zone was assumed that surrounded the urban areas. Rural residential centres were assumed to be located in the centre of the rural zone and surrounded by non-residential areas (agricultural land). Thus transport of compost produced from solid waste in urban centres to the outer border of urban agricultural land (short arrow) is included in the operational costs. The same model was used to determine the costs for recovered struvite transport from the urban residential areas to the middle of the rural area (long arrow). The transportation costs were set as 0.70 US$/km (Suletra et al., 2009) after correction for increased fuel prices and wages since 2009.

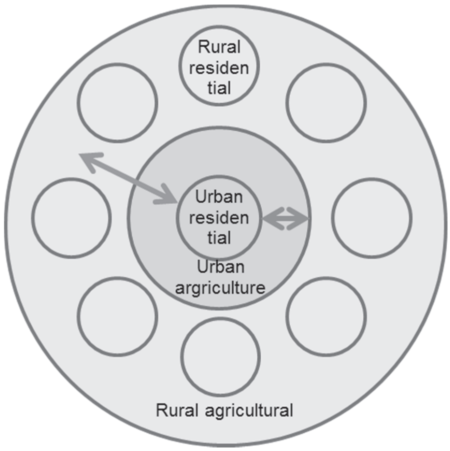


Figure A5.1 Applied transportation model

Section 4 Applied population and area forecasts data

In Section 3, step 4 of Chapter 5, we refer to applied population growth data and residential are faction of the total administrative units. The population forecast multipliers per indicated time interval are presented in Table A5.3.

Table A5.3 Applied population forecast multipliers per Province (BPS, 2013) to be multiplied by the 2010 population size of each administrative unit

Province number BPS code	Abbreviated Province name	2015		2020		2025		2030		2035	
		urban	rural	urban	rural	urban	rural	urban	rural	urban	rural
11	Aceh	1.19	1.08	1.41	1.14	1.66	1.17	1.92	1.17	2.20	1.16
12	SumUt	1.14	1.01	1.29	0.98	1.43	0.93	1.57	0.86	1.70	0.78
13	SumBar	1.22	0.98	1.45	0.93	1.67	0.88	1.88	0.82	2.08	0.75
14	Riau	1.16	1.14	1.32	1.27	1.48	1.39	1.64	1.51	1.80	1.62
15	Jambi	1.14	1.08	1.29	1.15	1.44	1.20	1.59	1.23	1.73	1.25
16	Sumsel	1.10	1.07	1.20	1.12	1.29	1.16	1.37	1.19	1.44	1.20
17	Bengkulu	1.10	1.09	1.22	1.16	1.34	1.21	1.45	1.26	1.56	1.29
18	Lampung	1.17	1.03	1.36	1.04	1.55	1.02	1.76	0.99	1.97	0.93
19	BaBel	1.19	1.06	1.40	1.08	1.63	1.08	1.87	1.06	2.13	1.01
21	KepRi	1.18	1.17	1.34	1.30	1.51	1.41	1.68	1.49	1.87	1.56
31	DKI	1.06	0.00	1.11	0.00	1.15	0.00	1.18	0.00	1.19	0.00
32	JaBar	1.24	0.81	1.43	0.68	1.60	0.57	1.74	0.48	1.86	0.39
33	JaTeng	1.20	0.93	1.31	0.91	1.43	0.88	1.55	0.83	1.66	0.78
34	Yogya	1.17	0.88	1.30	0.80	1.43	0.72	1.54	0.64	1.65	0.56
35	JaTim	1.23	0.90	1.36	0.85	1.48	0.79	1.60	0.73	1.70	0.65
36	Banten	1.15	1.08	1.30	1.11	1.49	1.05	1.69	0.90	1.93	0.68
51	Bali	1.16	0.93	1.31	0.84	1.45	0.76	1.58	0.68	1.70	0.60
52	NTB	1.14	1.02	1.32	1.01	1.50	0.97	1.69	0.91	1.88	0.83
53	NTT	1.21	1.06	1.48	1.11	1.79	1.15	2.16	1.18	2.59	1.18
61	Kalbar	1.18	1.05	1.38	1.07	1.61	1.07	1.85	1.05	2.10	1.00
62	Kalteng	1.23	1.07	1.50	1.13	1.81	1.15	2.14	1.15	2.50	1.12
63	Kalsel	1.18	1.04	1.36	1.06	1.56	1.05	1.76	1.02	1.96	0.96
64	Kaltim	1.21	1.04	1.42	1.06	1.63	1.07	1.85	1.04	2.07	0.99
71	Sulut	1.16	0.98	1.34	0.93	1.50	0.87	1.67	0.79	1.82	0.69
72	Sulteng	1.22	1.05	1.47	1.08	1.76	1.09	2.08	1.08	2.45	1.04
73	Sulsel	1.18	0.99	1.37	0.96	1.57	0.91	1.78	0.84	1.97	0.77
74	Sultra	1.29	1.06	1.60	1.10	1.96	1.12	2.34	1.12	2.77	1.10
75	Gorontalo	1.26	1.00	1.53	0.99	1.81	0.96	2.09	0.93	2.38	0.86
76	Sulbar	1.12	1.10	1.24	1.21	1.35	1.31	1.46	1.41	1.56	1.51
81	Maluku	1.17	1.06	1.30	1.13	1.44	1.20	1.58	1.26	1.72	1.31
82	Maluku Utara	1.24	1.08	1.40	1.18	1.56	1.27	1.72	1.35	1.89	1.43
91	Papua Barat	1.23	1.11	1.50	1.20	1.81	1.28	2.15	1.33	2.54	1.36
94	Papua	1.23	1.07	1.47	1.12	1.74	1.16	2.04	1.17	2.37	1.15
Indonesia		1.19	0.97	1.34	0.96	1.49	0.93	1.64	0.88	1.78	0.83

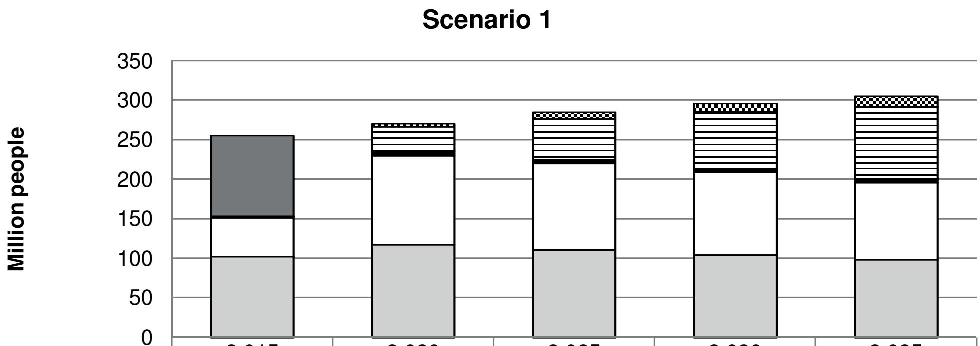
The average residential fraction of the total administrative area (e.g. corrected for agricultural use age) for Java island was based on the Java Spatial Model (MoPW, 2011) and all other islands are shown the determination of residential fractions per urban or rural and city or regency levels are shown in Table A5.4 The JSM shows the fraction of total land area used for residential purposes in the period 2015-2035 for urban and rural areas. These fractions were extrapolated to other Indonesian regions, applying a correction factor of 0.75 to compensate for these less densely populated areas.

Table A5.4 Ratio residential area/total area for four types of residential features (urban and rural and city (kota) and regencies (kabupaten). The upper part of the Table shows the Java values based on Java Spatial Model (JSM); the lower part of the Table shows the values for other islands applying the correction factor of 0.75

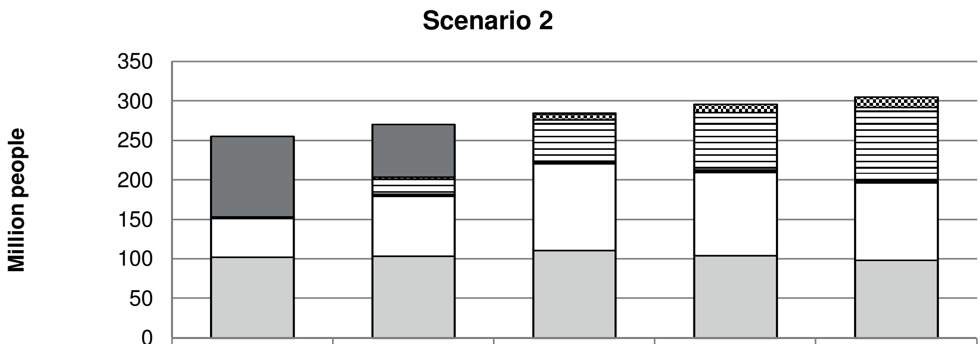
Region	Urban or rural / Administrative type	Urban or rural areas	
		urban	rural
Java (based on JSM)	City (Kota)	0.84	0.17
	Regency (Kabupaten)	0.37	0.16
Non-Java	City (Kota)	0.63	0.13
	Regency (Kabupaten)	0.28	0.12

Section 5 Numeric values of people served by WWT and MSW facilities

In paragraph 5.4.2 (Figures 5.2A/B; 5.3A/B) of Chapter 5 the development between 2015-2035 of people connected to a type of WWT system or served by a type of MSW system were presented. Figure A5.2 A/B and Figure A5.3 A/B below also show the numeric values of these developments



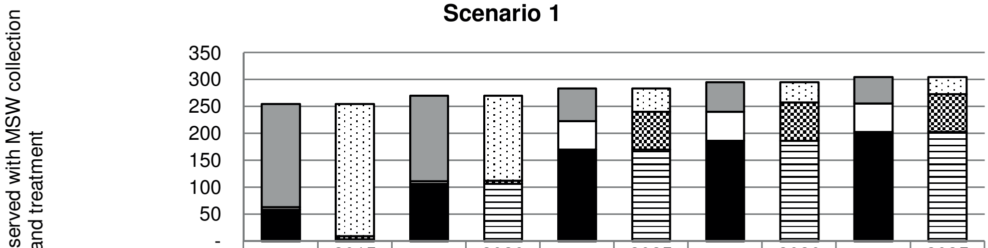
	2,015	2,020	2,025	2,030	2,035
Unimproved	102.051	-	-	-	-
Cen	.204	4.566	7.572	10.505	13.317
MedCen	.204	31.325	52.130	72.433	91.883
CBS	1.193	4.889	4.749	4.494	4.20
On-site rural	49.491	112.442	109.059	104.119	97.828
On-site urban	102.318	117.845	111.318	104.853	98.425



	2015	2020	2025	2030	2035
Unimproved	102.051	66.911	.679	-	-
Cen	.204	2.566	7.572	10.505	13.317
MedCen	.204	19.256	52.130	72.433	91.883
CBS	1.193	2.728	3.717	3.711	3.644
On-site rural	49.491	75.651	109.413	104.902	98.384
On-site urban	102.318	103.953	111.318	104.853	98.425

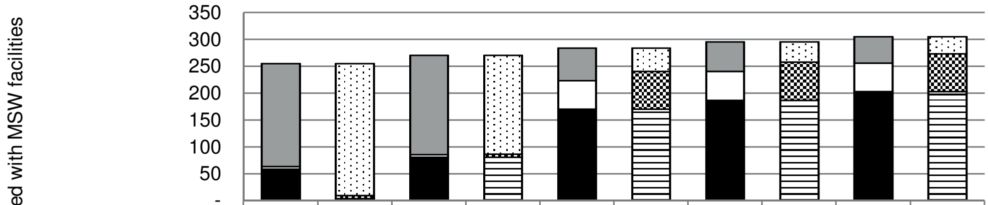
Figure A5.2 A (scenario 1; above) and **B** (scenario 2; below) WWT access development with urban and rural on-site systems, CBS and MedCen (medium-centralized) and Cen (centralized) systems

Scenario 1



		2015		2020		2025		2030		2035
Treatment not served		245.72		158.05		43.83		38.17		31.25
Treatment rural		5.68		5.68		69.97		70.44		70.78
Treatment urban		4.06		107.34		171.02		187.79		203.62
Collection not served	191.42		158.94		60.93		55.54		48.82	
Collection rural	5.01		5.01		52.87		53.07		53.21	
Collection urban	59.03		107.12		171.02		187.79		203.62	

Scenario 2



		2015		2020		2025		2030		2035
Treatment not served		245.72		183.56		43.83		38.17		31.25
Treatment rural		5.68		5.68		69.97		70.44		70.78
Treatment urban		4.06		81.82		171.02		187.79		203.62
Collection not served	191.42		184.59		60.93		55.54		48.82	
Collection rural	5.01		5.01		52.87		53.07		53.21	
Collection urban	59.03		81.47		171.02		187.79		203.62	

Figure A5.3 A (scenario 1; above) and **B** (scenario 2; below) MSW access development for collection and treatment services

Section 6 Examples of translation of national budgets to provincial budgets

In Chapter 5 (paragraph 5.4.2), national sanitation developments (e.g. budgets, systems) were provided. Because the framework was developed using information from the smallest available administrative units, the presented nationwide results can also be broken down to lower levels of governments. Below three examples of a “translation” of national planning to provincial levels for WWT and MSW budgets (Table A5.5), systems (Table A5.6) and percentage of investments contributing to poor households (Table A5.7) for the period 2015-2019 are presented. The break down provides insight in how budgets should be allocated to the different provinces to implement the planned national targets (Table A5.5). Also it provides useful information on the number and type of facility or system required in each province (Table A5.6) and how much of the investments will benefit poor households (Table A5.7)

Table A5.5 Provincial MSW and WWT investments (million US$) for 2015-2019: Scenario 1 [a]

prov code (BPS)	Province name	WWT 2015-2019					MSW 2015-2019			TOTAL
		On-site	CBS	Medium central	Central	Total WWT	Collec-tion	Treat-ment	Total MSW	WWT & MSW
11	Aceh	317	21	131	-	470	6	-	6	476
12	SumUt	533	21	418	216	1,188	35	372	407	1,595
13	SumBar	275	1	102	27	404	12	-	12	416
14	Riau	319	6	80	12	417	9	82	90	508
15	Jambi	191	6	50	-	247	4	42	45	292
16	Sumsel	500	16	121	81	717	11	96	108	825
17	Bengkulu	96	4	47	-	147	1	18	19	166
18	Lampung	354	20	140	42	557	10	56	67	624
19	BaBel	47	-	24	3	73	4	-	4	78
21	KepRi	55	3	56	9	123	4	68	71	195
31	DKI	3	-	476	193	672	-	426	426	1,098
32	JaBar	677	57	3,455	509	4,698	104	1,404	1,508	6,206
33	JaTeng	839	149	636	-	1,624	52	596	649	2,273
34	Yogya	27	0	72	-	100	6	76	81	181
35	JaTim	1,232	198	1,095	121	2,645	68	972	1,039	3,685
36	Banten	328	91	631	32	1,082	29	405	434	1,515
51	Bali	51	0	104	14	168	8	49	57	226
52	NTB	161	45	219	37	462	12	112	124	585
53	NTT	358	7	128	22	514	6	27	34	548
61	Kalbar	309	-	72	12	393	11	-	11	404
62	Kalteng	214	0	28	-	243	4	-	4	247
63	Kalsel	242	3	74	9	327	8	79	86	414
64	Kaltim	142	3	99	80	324	6	39	45	369
71	Sulut	61	1	65	40	167	4	-	4	171
72	Sulteng	137	2	41	6	187	4	-	4	191
73	Sulsel	220	9	151	113	493	11	17	27	520
74	Sultra	115	4	55	-	175	2	-	2	177
75	Gorontalo	58	1	40	-	99	3	17	20	119
76	Sulbar	70	8	15	-	93	1	-	1	94
81	Maluku	97	1	18	14	130	2	-	2	132
82	Maluku Utara	65	4	21	-	89	1	-	1	91
91	Papua Barat	82	8	31	-	121	3	-	3	124
94	Papua	441	1	46	-	488	5	38	43	531
	TOTAL	**8,616**	**690**	**8,742**	**1,592**	**19,640**	**446**	**4,991**	**5,437**	**25,077**

[a] The total of 25.1 billion US$ (right lower corner) compared to the 25.2 billion US$ mentioned in Chapter 5 (Table 5.9) is the result of intermediate rounding

Table A5.6 Provincial new MSW and WWT facilities for 2015-2019: Scenario 1 [a, b]

prov code (BPS)	Province name	WWT 2015-2019					MSW 2015-2019			TOTAL
		On-site	CBS	Medium central	Central	Total WWT	Collection	Treatment	Total MSW	WWT & MSW
11	Aceh	317	21	131	-	470	6	-	6	476
12	SumUt	533	21	418	216	1,188	35	372	407	1,595
13	SumBar	275	1	102	27	404	12	-	12	416
14	Riau	319	6	80	12	417	9	82	90	508
15	Jambi	191	6	50	-	247	4	42	45	292
16	Sumsel	500	16	121	81	717	11	96	108	825
17	Bengkulu	96	4	47	-	147	1	18	19	166
18	Lampung	354	20	140	42	557	10	56	67	624
19	BaBel	47	-	24	3	73	4	-	4	78
21	KepRi	55	3	56	9	123	4	68	71	195
31	DKI	3	-	476	193	672	-	426	426	1,098
32	JaBar	677	57	3,455	509	4,698	104	1,404	1,508	6,206
33	JaTeng	839	149	636	-	1,624	52	596	649	2,273
34	Yogya	27	0	72	-	100	6	76	81	181
35	JaTim	1,232	198	1,095	121	2,645	68	972	1,039	3,685
36	Banten	328	91	631	32	1,082	29	405	434	1,515
51	Bali	51	0	104	14	168	8	49	57	226
52	NTB	161	45	219	37	462	12	112	124	585
53	NTT	358	7	128	22	514	6	27	34	548
61	Kalbar	309	-	72	12	393	11	-	11	404
62	Kalteng	214	0	28	-	243	4	-	4	247
63	Kalsel	242	3	74	9	327	8	79	86	414
64	Kaltim	142	3	99	80	324	6	39	45	369
71	Sulut	61	1	65	40	167	4	-	4	171
72	Sulteng	137	2	41	6	187	4	-	4	191
73	Sulsel	220	9	151	113	493	11	17	27	520
74	Sultra	115	4	55	-	175	2	-	2	177
75	Gorontalo	58	1	40	-	99	3	17	20	119
76	Sulbar	70	8	15	-	93	1	-	1	94
81	Maluku	97	1	18	14	130	2	-	2	132
82	Maluku Utara	65	4	21	-	89	1	-	1	91
91	Papua Barat	82	8	31	-	121	3	-	3	124
94	Papua	441	1	46	-	488	5	38	43	531
	TOTAL	**8,616**	**690**	**8,742**	**1,592**	**19,640**	**446**	**4,991**	**5,437**	**25,077**

[a] Compared to the numbers presented in Table 5.8 of Chapter 5 only the new facilities are required

[b] Compared to the numbers presented in Table 5.8 of Chapter 5 also MSW collection vehicles, transport trucks are presented. For specifics on the calculation of these vehicles reference is made to Kerstens et al. (2015)

Table A5.7 Calculated percentage of provincial investments per type of WWT systems in the period 2015-2019 benefiting poor people, following Scenario 1

Province name	WWT 2015-2019				
	On-site	CBS	Medium central	Central	Total WWT
Aceh	30%	33%	18%	0%	27%
SumUt	19%	20%	12%	9%	15%
SumBar	18%	20%	14%	15%	17%
Riau	15%	13%	13%	17%	15%
Jambi	17%	25%	20%	0%	18%
Sumsel	19%	21%	18%	20%	19%
Bengkulu	28%	32%	17%	0%	24%
Lampung	30%	30%	24%	22%	28%
BaBel	19%	0%	21%	25%	20%
KepRi	18%	19%	17%	19%	18%
DKI	21%	0%	17%	16%	17%
JaBar	29%	34%	17%	12%	19%
JaTeng	36%	42%	26%	0%	33%
Yogya	47%	52%	37%	0%	40%
JaTim	43%	46%	28%	18%	36%
Banten	19%	24%	11%	16%	15%
Bali	25%	15%	18%	11%	19%
NTB	49%	53%	43%	38%	46%
NTT	20%	20%	7%	10%	16%
Kalbar	15%	0%	15%	19%	15%
Kalteng	11%	12%	7%	0%	10%
Kalsel	17%	18%	10%	9%	15%
Kaltim	16%	28%	9%	7%	12%
Sulut	18%	20%	12%	7%	13%
Sulteng	24%	26%	9%	7%	20%
Sulsel	24%	27%	14%	9%	18%
Sultra	23%	27%	12%	0%	20%
Gorontalo	23%	26%	20%	0%	22%
Sulbar	20%	25%	17%	0%	20%
Maluku	18%	23%	13%	17%	17%
Maluku Utara	8%	10%	3%	0%	7%
Papua Barat	30%	25%	6%	0%	23%
Papua	34%	25%	9%	0%	31%
TOTAL	**27%**	**37%**	**19%**	**14%**	**23%**

Section 7 Example of a City Sanitation Strategy: Tegal

In Indonesia, cities must prepare a City Sanitation Strategy to be eligible for national funding. Thus, a 5-year plan, using a similar residential area-based sanitation system selection as presented in the framework, is prepared to formulate budgets and specify required institutional and advocacy and campaigning activities (Kearton et al., 2013; Parkinson et al., 2014; USDP, 2014). This simultaneous development of a top-down supply and bottom-up demand for sanitation funding links the Indonesian central government’s policy making and oversight role with the local governments’ role for implementation (ADB, 2013a).

An example of the outcome of a CSS for Tegal in Central Java is presented below. The CSS applies a similar residential area based system selection of WWT and MSW systems as presented in the planning framework. The CSS and the presented planning framework have both been developed in close cooperation with the Indonesian Ministry of Planning and Public Works and with support of the USDP (Urban Sanitation Development Program) project.

Tegal, located in Central Java, has been working on sanitation developments for several years, but with limited success. It has shown a weak institutional set-up, resulting in the absence of properly functional septic management system, a poorly designed sludge processing facility, limited demand for septage treatment services, poorly designed community based sewerage, and limited government control over the details used for leach-pits. The city of Tegal started with the preparatory works of a CSS in 2009. Following identified obstacles in implementation and renewed interest by the City of Tegal, an updated CSS was prepared together with the Sanitation Working group of Tegal within the USDP framework in 2012 and 2013.

To allow for prioritization of implementation, the CSS starts with a risk area analysis, based on the impact that lack of WWT and MSW services may have on the type of area (as a function of population size, density, urban and rural functions and poverty level) (Parkinson et al., 2014). This risk map is prepared on a *desa/ kelurahan* level (smallest administrative unit) for both wastewater and solid waste (Figure A5.4).

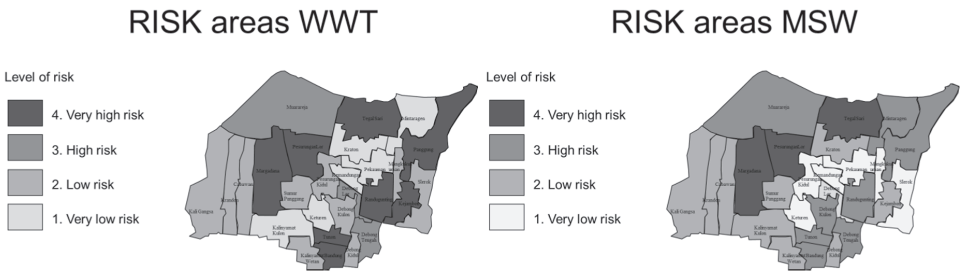


Figure A5.4 Risk analysis for WWT and MSW situation in Tegal

Following the urban/rural features and population density a selection of wastewater and solid waste systems zones on a *desa/ kelurahan* level is prepared Figure A5.5 & Table A5.8 for WWT and Figure A5.6 and Table A5.9 for MSW. In Figure A5.5 medium centralized WWT are presented as IPAL kawasan, which is the Indonesian name. The level of priority is based on the risk analysis (Figure A5.4).

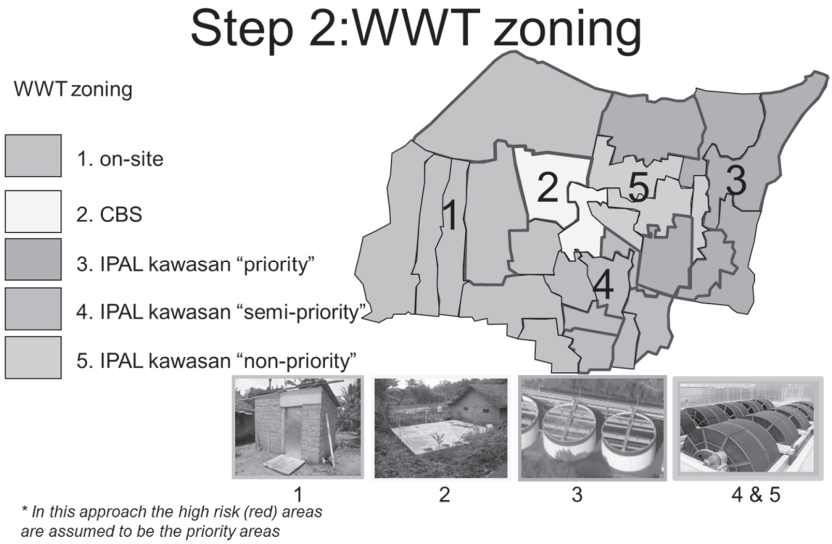


Figure A5.5 WWT zoning for Tegal city following population density and urban/rural features

Table A5.8 WWT implementation for Tegal

Item	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5
Type of system	on-site system (shared or individual)	Hybrid: Community Based System Communal septic tank, IPAL-communal or MCK+ with small sewer	Off-site: semi-centralised system	Off-site: semi-centralised system	Off-site: semi-centralised system
Type of sewer system technology	no sewer system	community sewer system	(pumped) sanitary sewer system	(pumped) sanitary sewer system	(pumped) sanitary sewer system
Type of WWTP technology	On-site: ind. septic tank	Hybrid: CBS - IPAL communal	Off-site: Aerobic activated sludge system	Off-site: Rotating Bio Contactor	Off-site: Rotating Bio Contactor
Number of total treatment systems required	10,333	28	2	2	2
Number of systems to be completed 1-5 years	5,741	14	2	0	0
Number of systems to be completed 6-10 years	4,593	14	0	2	0
Number of systems to be completed 11-20 years	0	0	0	0	2

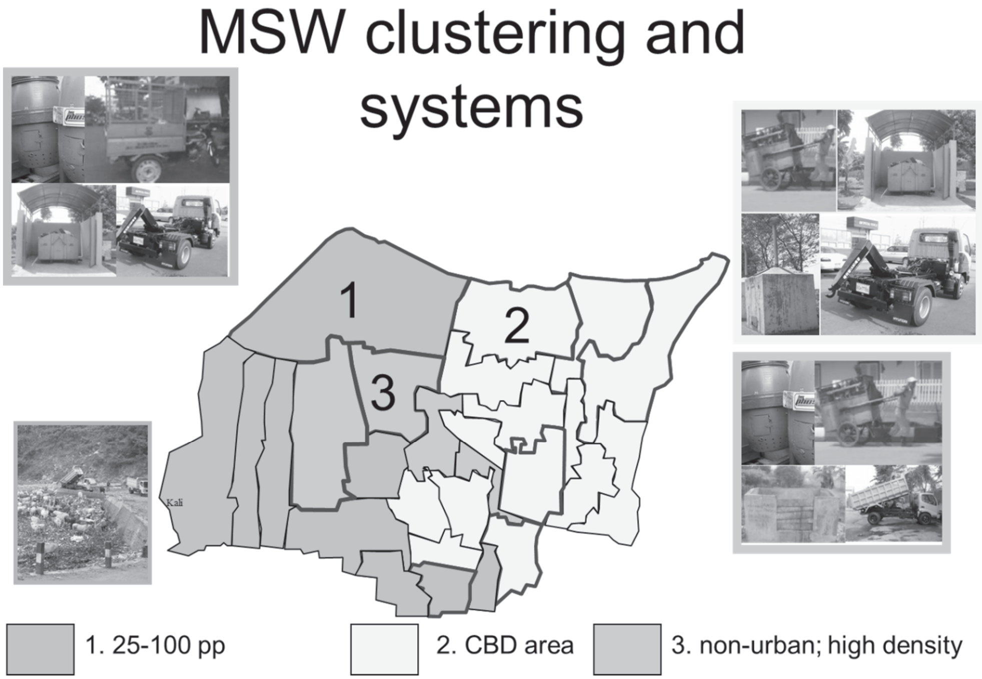


Figure A5.6 MSW zoning for Tegal city following population density and urban/rural features. Zone 1 applies home composting followed by a collection using motorized vehicles to a covered intermediate transfer station. In Zone 2 mixed household waste is collected by hand carts and transferred to a transfer station with decentralized 3R. From the transfer stations, remaining waste of Zone 1 and 2 is transported by arm roll truck to a centralized landfill. In Zone 3, home composting followed by collection using hand carts to small scale transfer plants was selected. Dump trucks were selected to transfer the waste from these transfer stations to the landfill

Table A5.9 MSW implementation for Tegal city

Zone number	Zone 1	Zone 2	Zone 3
Is 3R promoted	yes	yes	yes
Collection system			
Proposed type of collection system	motorized car	hand pulled cart	hand pulled cart
Number of collection systems required until 5 y	10	31	4
Number of collection systems required until 20 y	17	61	9
Transfer station			
Proposed type of transfer system	Transfer Depo III + container	Transfer Depo III + container	TPS-biasa
Number of new transfer station until 5 y	3	2	4
Number of new transfer station until 20 y	4	4	8
Transport system			
Number of new transport trucks until 5 y			
dump trucks	0	0	1
armroll truck	2	2	0
Number of new transport trucks until 20 y			
dump trucks	0	0	2
armroll truck	2	3	0
Number of container trucks until 20 years			
Treatment (composting+plastic recovery)			
Level of treatment	household level	Transfer Depo level	household level
Number of composter units until 5y	4000	9	1250
Number of composter units until 20y	6800	18	3200

The development of the population with access to wastewater (Figure A5.7) and solid waste services (Figure A5.8) is determined based on (1) set targets by the local government and (2) the residential area based system selection.

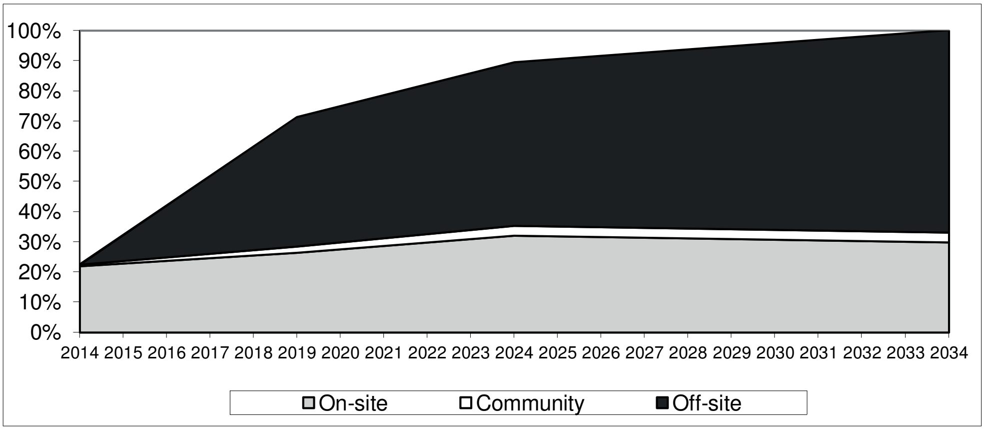


Figure A5.7 Tegal population development with access to indicated type of wastewater treatment system

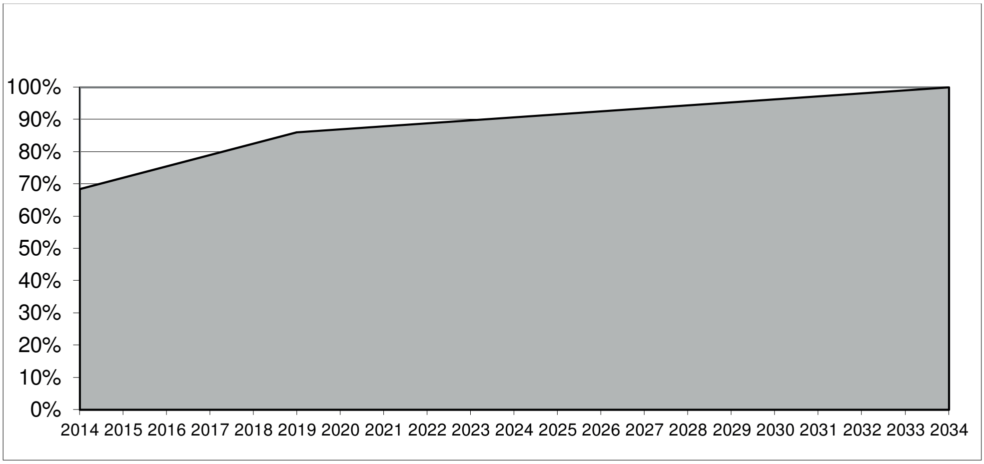


Figure A5.8 Tegal population development with access to MSW services

The temporal investments for each of these activities are then plotted. Similar to the planning framework it distinguishes software (institutional strengthening, campaigning & advocacy to assure sustainable operation), hardware and land (site) costs. The investment schedules for wastewater and solid waste are shown in Figure A5.11 and Figure A5.12 respectively.

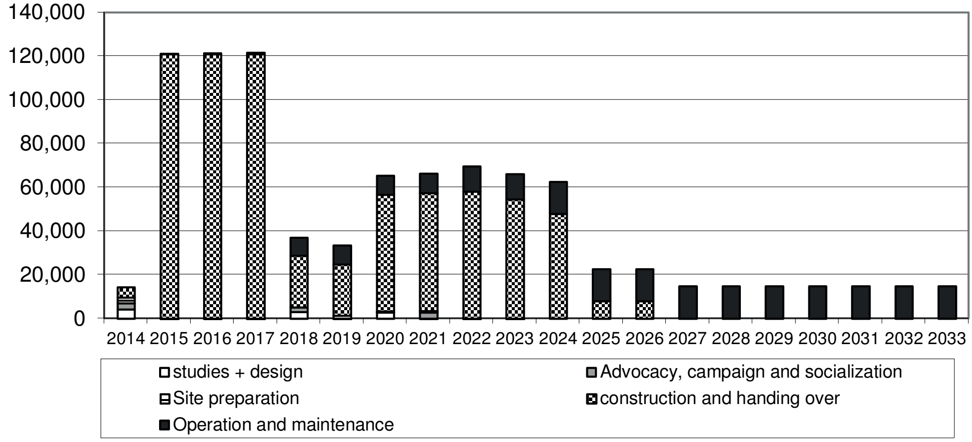


Figure A5.9 Tegal WWT investment and operational costs per year in million Rp (1 US$ = 10,000 Rp)

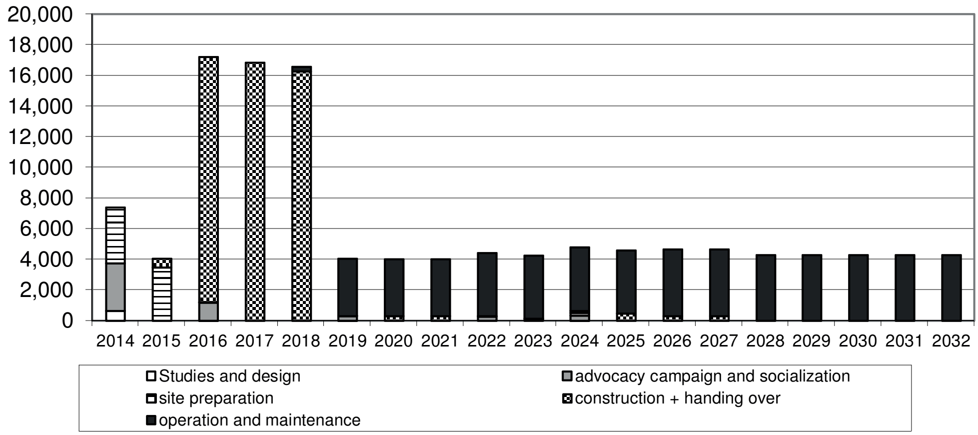


Figure A5.10 Tegal MSW investment and operational costs per year in million Rp (1 US$ = 10,000 Rp)

Finally, for the first 5 year a detailed budget break-down per zone and activity is prepared. These are presented (in million Rp) in Figure A5.11 (WWT) and Figure A5.12(MSW).

Summary of activities and costs

	2014	2015	2016	2017	2018	
	million Rp	million Rp	million Rp	million Rp	million Rp	
Citywide (inc IPLT)						
Masterplan	1,200	-	-	-	-	
Studies and design for IPLT	-	-	-	-	-	
Advocacy, campaign and socialization for IPLT	200	-	-	-	-	
Site preparation IPLT	-	-	-	-	-	
Construction, supervision and procurement IPLT	-	-	-	-	-	
Operation and Maintenance IPLT	-	-	131	131	131	
zone 1						
Studies and design	-	-	-	-	-	
Advocacy, campaign and socialization	178	178	178	178	178	
Site preparation	-	-	-	-	-	
Construction, supervision and handing over	3,444	3,444	3,444	3,444	3,444	
Operation and Maintenance	-	172	344	517	689	
zone 2						
Studies and design	35	35	35	35	35	
Advocacy, campaign and socialization	75	75	75	75	75	
Site preparation	42	42	42	42	42	
Construction, supervision and handing over	1,300	1,300	1,300	1,300	1,300	
Operation and Maintenance	-	30	60	90	120	
zone 3						
Studies and design	4,200	-	-	-	-	
Advocacy, campaign and socialization	2,350	-	-	-	-	
Site preparation	2,640	-	-	-	-	
Construction, supervision and handing over	-	115,840	115,840	115,840	18,807	
Operation and Maintenance	-	-	-	-	7,440	
zone 4						
Studies and design	-	-	-	-	3,000	
Advocacy, campaign and socialization	-	-	-	-	1,870	
Site preparation	-	-	-	-	-	
Construction, supervision and handing over	-	-	-	-	-	
Operation and Maintenance	-	-	-	-	-	
zone 5						
Studies and design	-	-	-	-	-	
Advocacy, campaign and socialization	-	-	-	-	-	
Site preparation	-	-	-	-	-	
Construction, supervision and handing over	-	-	-	-	-	
Operation and Maintenance	-	-	-	-	-	
TOTAL in first 5 years (inc O&M)	15,664	121,116	121,450	121,652	37,131	417,013

Figure A5.11 Tegal WWT 5 year investment per identified zone

Summary of activities and costs of first five years	2014	2015	2016	2017	2018	
Citywide	million Rp	million Rp	million Rp	million Rp	million Rp	
Masterplan	-	-	-	-	-	
Studies and design	600	-	-	-	-	
advocacy campaign and socialization	1,350	-	-	-	-	
site preparation	3,500	3,500	-	-	-	
construction + handing over	-	-	15,450	15,450	15,450	
operation and maintenance	-	-	-	-	-	
Cluster 1						
Studies and design	-	-	48	-	-	
advocacy campaign and socialization	-	-	1,150	-	-	
site preparation	-	-	30	-	-	
construction + handing over	-	-	-	847	847	
operation and maintenance	-	-	-	-	-	
Cluster 2						
Studies and design	67	-	-	-	-	
advocacy campaign and socialization	600	-	-	-	-	
site preparation	146	-	-	-	-	
construction + handing over	-	429	429	429	-	
operation and maintenance	-	-	-	-	115	
Cluster 3						
Studies and design	-	-	-	-	-	
advocacy campaign and socialization	1,150	-	-	-	-	
site preparation	10	-	-	-	-	
construction + handing over	-	130	130	130	-	Total
operation and maintenance	-	-	-	-	188	62,176

Figure A5.12 Tegal MSW 5 year investment per identified zone

Section 8 References Appendix Chapter 5

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Chapter 6

Potential demand for recoverable resources from Indonesian wastewater and solid waste

This chapter has been accepted for publication in a slightly modified version in *Resources, Conservation & Recycling* as Potential demand for recoverable resources from Indonesian wastewater and solid waste

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Abstract

Projected population growth and urbanization will become a challenge for finite natural resources, their distribution and local availability. At the same time, 2.5 billion people do not have access to sanitation facilities. Indonesia is one of these rapidly growing countries with a poorly developed municipal wastewater and solid waste sector. Without an integrating concept to recover and reuse resources, "waste flows" are discarded and their potential value is ignored. Therefore, the Indonesian backlog may be an opportunity, since it allows for direct introduction of a circular resource approach. To foster a sustainable municipal wastewater and solid waste management, the 20 years' demand forecast of recoverable resources (phosphorus, compost, duckweed, plastic and paper) was analyzed. Phosphorus, compost and duckweed analysis was based on *nutritional* demand and not on *market* demand. Demand for recoverable plastic and paper related to the potential substitution of conventionally manufactured products. Phosphorus and compost demand analysis was based on (1) fertilizer requirements of 68 crops (staple food, horticulture and plantation), and (2) anticipated increase in production area of these crops. Duckweed demand as a protein-rich fish feed was analyzed based on the forecasted demand from aquaculture (tilapia and carp). The potentially recoverable (waste) plastic and paper to substitute conventional manufactured products were based on extrapolation of past trends in plastic and paper production in Indonesia. The potential contribution of recoverable products to the forecasted demand for 2035 was assessed for phosphorus (15%), compost (35%), duckweed (7%), plastic (66%) and paper (18%). A geographical discrepancy between potential recovery and demand location for phosphorus and compost was found. Therefore, the locations of potential markets should be considered in the planning and selection of wastewater and solid waste facilities. The presented methodology to assess the potential demand for recoverable resources from wastewater and solid waste may be applied in other countries as well.

Keywords: resource recovery; demand analysis; agriculture; wastewater; solid waste; phosphorus

6.1 Introduction

The coming decades will bring profound changes to the size and spatial distribution of the global population. The continuing urbanization and overall growth is projected to add 2.5 billion people to the urban population by 2050, with nearly 90% of the increase concentrated in Asia and Africa (United Nations, 2014). In the coming 20 years Indonesia's population is expected to grow from the current 250 million to over 305 million people (BPS, 2013). By 2035 an estimated two third of the Indonesian population will live in urban areas compared to the current 50% (BPS, 2013). These developments pose a challenge for food security and represent additional pressure on the food system (production and related supply of commodities) and on finite natural resources, their distribution and local availability (Thornton, 2010; Gerbens-Leenes et al., 2010; Cordell et al., 2011).

Increasing urbanization is also a challenge for the Indonesian sanitation sector that, despite modest improvement over the past years is still in a poor state (ADB, 2013; Kearton et al., 2013; WHO & UNICEF, 2014). Indonesia is not an exception, since worldwide 2.5 billion people do not have access to an improved sanitation facility and some 80 countries are not on track or made insufficient progress to achieve the Millennium Development Goals on sanitation (WHO & UNICEF, 2014). However, abundance of unmanaged solid waste and wastewater may result in an abundance of food and economic growth if resources (e.g. phosphorus (P), organic fertilizer, plastics) are managed, reused and recovered (Mcdonough & Braungart, 2000; Braungart et al., 2007; McDonough & Braungart, 2010; Kerstens et al., 2011). Therefore, an aspect of wastewater and solid waste development that receives increasing attention is the potential to recover resources from wastewater and solid waste (Lettinga, 2006; Almy, 2008; Aprilia et al., 2012; Thibodeau et al., 2014). Besides the needs of households provided with sanitation systems ("front-end" users), also the needs of potential users of sanitation products ("back-end" user) should be considered to foster long-term operational and financial sustainability (Murray & Ray, 2010). Back-end users comprise among others agriculture (Schröder et al., 2011), horticulture (Aye & Widjaya, 2006), aquaculture (Mungkung et al., 2013) and plastic and paper processing industries (APKI, 2012; GBGIndonesia, 2013). Indonesia aims to have universal sanitation access by 2019 and massive implementation of wastewater and solid waste treatment facilities are planned (Bappenas, 2014). Therefore, the backlog in development of these facilities could be an opportunity. Including the concept of resource recovery in the planning allows for direct introduction of a circular resource management, instead of developing a linear management system (Agudelo-Vera et al., 2011).

To assure a favorable financial perspective, it is essential to understand the demand for recoverable resources. Demand is affected by (1) population developments (BPS, 2013), (2) availability of resources (Cordell et al., 2011), (3) quality of produced products (Snyman & Vorster, 2011), (4) efficiencies of systems (Gerbens-Leenes et al., 2010), (5) recovery costs compared to prices of competitive resources (Saveyn & Eder, 2014), and (6) existing or planned policies and frameworks (WHO, 2006a; Cordell et al., 2011). Kerstens et al. (2015) demonstrated the impact of varying selling price of recovered resources on the financial attractiveness of resource recovery from wastewater and solid waste. Studies have related resource recovery potential with demand for a community or region using “Material Flow Analysis” (Meinzinger et al., 2009; Ushijima et al., 2012), applying a sensitivity analysis or optimization to understand the balance between supply and demand (Friedler, 2004; Kerstens et al., 2009) or by comparing global resource recovery potential with global demand (De Graaff et al., 2011; Mihelcic et al., 2011). Despite differences (e.g., resource, study location, or type of applied technologies) these studies quantify and qualify demand of recoverable resources and allow a policy maker to make informed choices on the feasibility of resource recovery (Ward, 2012).

The government of Indonesia is preparing the wastewater and solid waste plans for the period 2015 to 2019 (Bappenas, 2014). In these plans a 20% reduction of solid waste being usually landfilled through 3R (Reduce, Reuse and Recycle) programs is formulated (MoPW, 2013; Bappenas, 2014). However, the demand for recovered resources is not included in the planning and the reuse potential is not clear.

In this study the potential demand for resources that can be recovered from municipal wastewater and solid waste in Indonesia is analyzed. Starting from past production and consumption patterns, an assessment for future demand is derived. This potential demand is then compared to the amount of resources that could be recovered. The selection of studied recoverable resources is based on four criteria:

(1) Resources should have a predicted increased demand in the future;
(2) Systems to recover or produce these resources should fit the Indonesian context (Kerstens et al., 2015);
(3) Recovered resources should be solid or stackable to allow for energy efficient transport. As a reference a dry matter content of at least 5 % is assumed;
(4) Recovery or production of resources from solid waste is restricted to the three largest fractions (organic waste, plastic and paper) in solid waste.

Table 6.1 presents the selected resources and the form in which they can be recovered in relation to these four criteria.

Table 6.1 Applied criteria for selection of resources that can be recovered or produced from wastewater and solid waste, and the form in which these can be recovered

Resource	Criteria				Form of recovery
	1. Resource demand in future	2. Feasible resource recovery technologies	3. Solid or stackable resources	4. Fraction in solid waste	
Phosphorus	Essential and scarce plant nutrient without substitute in food production. Demand will increase [a]	Wastewater, side stream of digester supernatant and sludge processing [b]	Struvite, Calcium phosphate > 90% [c], Compost > 65% [d]; Fresh Duckweed > 5% [e]	In organic fraction	Struvite [b, c]; Compost [f]; Duckweed [e]
Organic matter	Maintain soil organic content . Demand will increase [g]	Composting of solid waste and sludge [f]	Compost > 65% [d];	Organic content: 59% [h].	Compost [f]; Duckweed [e]
Proteins	Demand as fish feed stock with high fish production growth rates [i]	Production as duckweed in ponds [e]	Duckweed > 5%; typical protein content is 20% of dry matter [e]	Not applicable	Duckweed [e]
Plastic and paper	Large increase in demand [j]	Recoverable by separation and processing [k]	Easily stackable [k]	Plastic: 14%; Paper: 12%	Plastic and paper waste

[a] Janssen et al. (1990), Syers et al. (2008), Cordell et al. (2011); [b] Kerstens et al. (2015) , Cornel & Schaum (2009), De Graaff et al. (2011), Le Corre et al. (2009), Egle et al. (2015); [c] Giesen (1999); [d] Kerstens et al. (2015), Hamelers (2001); [e] El-Shafai et al. (2007); [f] Kerstens et al. (2015), Veeken (2005), Veeken et al. (2003), Koné et al. (2007), [g] Smaling & Janssen (1993), Minasny et al. (2011); [h] Aprilia et al. (2013); [i] FAO (2010), Journey et al. (1993); [j] GBGIndonesia (2013), Cornelia et al. (2013), Kemenperin (2012b), Handoyo (2014) ; [k] Sasaki & Araki (2013)

In this study recoverable resources are related to their potential practical use. Thus, organic matter is a resource, but demand in agriculture and recovery from wastewater and solid waste is analyzed as compost. Proteins demand in aquaculture (as fish feed) and recovery from wastewater will be studied as duckweed. Demand for P, compost and duckweed refer to *nutritional demand* for crops or fish production and not to *market demand* by their producers (e.g. farmers). The focus of the plastic and paper analysis is the recovery potential to substitute conventionally manufactured products by recyclables from municipal solid waste.

6.2 Materials and methods

Developments from the year 2000 in production of agriculture, horticulture, plantation and aquaculture in Indonesia were analyzed. Based on this data a forecast for the production until 2035 of staple food, horticulture and plantation crops as well as fish production was made. The current and future (potential) demand for a recoverable resource was analyzed for each crop or product. Plastic and paper past production and consumption patterns were collected and used to forecast future demand. The determined demand for P, compost, duckweed, plastic and paper was compared with the potential recovery from wastewater and solid waste (Figure 6.1). Demand analysis were based on secondary data made available through the Central Statistical Bureau (*Buro Pusat Statistik*; BPS) (BPS, 2014), Ministry of Agriculture (MoA) (MoA, 2014) and the Ministry of Industry (MoI) (Kemenperin, 2012b). Additional information was gathered through meetings with industrial associations (APKI, 2012), Universities (Padjajaran University, 2013) and FAO (2014b). Finally, an analysis of impact of different growth forecasts on the demand was made.

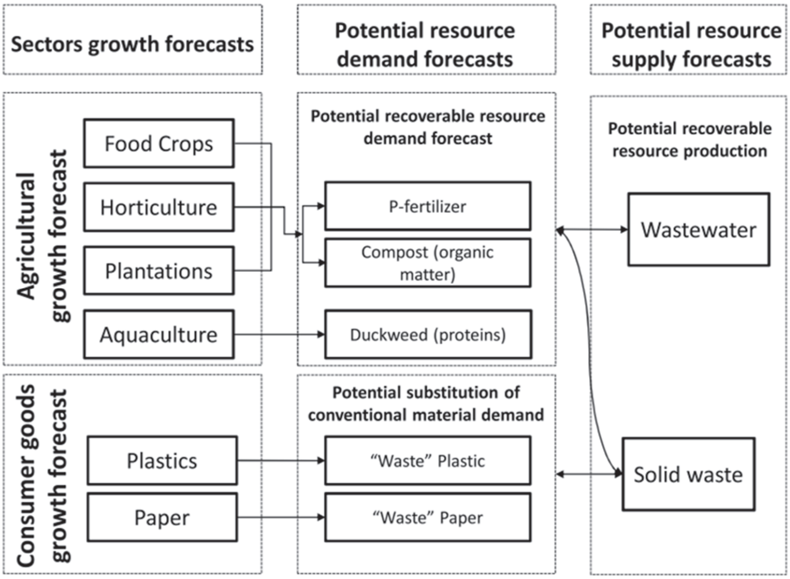

Figure 6.1 Schematic presentation of the framework for forecasting the recoverable resource demand compared to resource recovery from wastewater and solid waste

6.2.1 Fertilizer and compost demand

Fertilizer nitrogen (N), phosphorus (P) and potassium (K), and compost (organic matter) demand in agriculture were each determined by the same method consisting of eight consecutive steps (Figure 6.2).

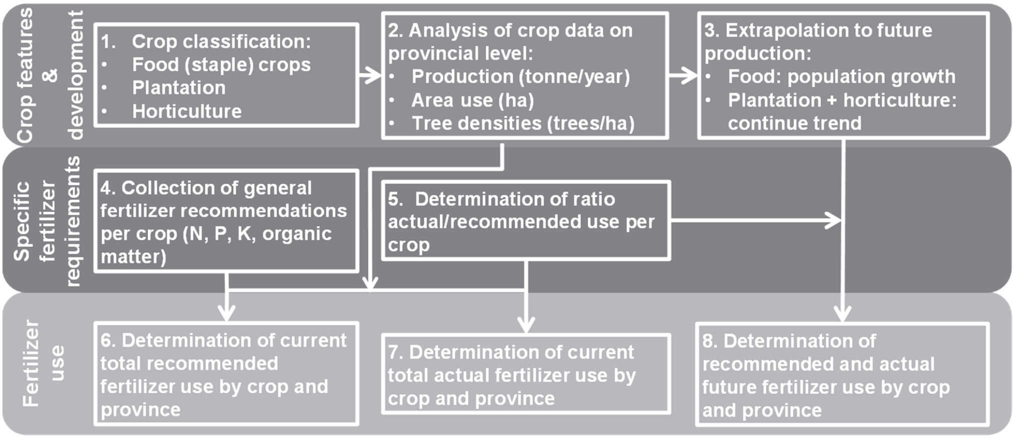

Figure 6.2 Flow chart showing the method of determination of future fertilizer use per crop and province in Indonesia

Step 1. *Crop classification:* Seven (staple) food crops, 15 plantation and 46 horticulture crops following the BPS website (BPS, 2014) were analyzed. Staple food crops comprised grains, pulses and root crops, including rice, maize, soybean, cassava, groundnut, green beans and sweet potato. Plantation crops include, among others, palm oil, sugar palm, rubber, coffee and tea (MoA, 2014) and are classified by the production management, making a distinction between small holder estate (SHE; 41%) and large estates (LE; 59%) (MoA, 2014). Horticulture crops comprised vegetables, fruits, spices, ornament and medicinal plants. Ornament and medicinal production amount to respectively 0.08% and 1% of the total horticultural area (MoA, 2014) and were therefore excluded. In the Appendix Chapter 6 (Section 1, Table A6.1) all analyzed crops are presented.

Step 2. *Analysis of crop features on provincial level*: Data on harvested area, tree densities per hectare (ha) for tree crops and type and quantity of crop production on a provincial level were collected (Appendix Chapter 6, Table A6.2). For food, plantation, and horticulture crops the latest available data base years were 2013, 2011 and 2012. Data on food crops was obtained from BPS (2014). Data on plantation and horticulture crops was derived from the Ministry of Agriculture (MoA, 2014).

Step 3. *Extrapolation of future crop production:* In the period 2000 to 2011 (staple) food production increased with 3% annually, whereas population increase was below 1.5% per year (BPS, 2014). In 2009 food import and export were respectively 8% and 10% of the food production (FAO, 2014b), indicating that most food produced in Indonesia is for the domestic market. For the 20 years projection, provincial food production increase is expected to follow provincial population increase (BPS, 2013). Between 2000 and 2011 the total plantation crop production in tonne (t) per year increased 6.6% annually of which the percentage of palm oil grew from nearly 40% to over 60%, corresponding with 11%

growth per year (MoA, 2014). Palm oil production projections until 2025 anticipate a reduced, but still high growth exceeding 7% annually (Bambang, 2011). The massive increase of especially the palm oil sector has resulted in environmental and land degradation and highlighted the need for a more sustainable approach in production (Harmen Smit et al., 2013; Yoshizaki et al., 2013). Therefore, the growth forecast until 2035 uses a linear forecast, corresponding with a reduced growth compared to an exponential one. Between 2000 and 2012 the horticultural sector showed an increase of 5.7 % in crop production (t/year) (MoA, 2014). Although horticulture largely concerns food products, the long term growth is expected to be higher than the explained (staple) food production forecast because of a change in diets with increasing income (Pingali, 2007). Therefore a linear trend based on the 2000-2012 was used to forecast the 2035 horticulture production. This resulted in a higher growth than staple food.

Step 4. *Collection of general fertilizer recommendations per crop:* For each of the 68 crops, data on recommended fertilizer application dosage per area or tree was collected. Besides nutrient removal by crops, fertilizer recommendation consider soil nutrient reserves, unavailability of the applied nutrients to the plant roots due to fixation, leaching or other losses (FAO & IFA, 2000). For the majority of food crops and plantation crops data were obtained through direct communication with the agricultural department of Padjajaran University in Bandung (Padjajaran University, 2013) and verified with the FAO guidelines (FAO, 2005). Horticultural fertilizer recommendations were obtained from the Ministry of Agriculture (MoA, 2012a), (MoA, 2012b) and literature (Weiss, 2002). As the guidelines of Padjajaran University, mention no specific requirements on the composition of organic matter, this study interprets organic matter as compost, complying with the national regulation (BSN, 2004). Fertilizer recommendations and tree densities are shown in Section 1 of the Appendix Chapter 6.

Step 5. *Comparison of recommended and actual fertilizer application:* Actual fertilizers application rates in the field by farmers vary and are generally less than recommended values as a result of insufficient information and financial means (Kariyasa, 2005; Irawan et al., 2012). Irawan et al. (2012) provide the percentage of actual inorganic fertilizer use compared to the recommended use for rice (68%), maize (37%) and soy bean (42%) for Indonesia. For other food crops and all horticultural crops no data on actual use was available. Therefore, the average values reported by Irawan et al. (2012) of 50% was used. Actual fertilizer use for plantation crops was based on Bambang (2011) who shows that 30% and 80% of SHE of oil palm and rubber plantations, respectively, apply fertilizers, whereas 100% of LE apply fertilizers. These percentages were interpreted as the fraction of recommended fertilizer use in this current study. Large discrepancies between the SHE and LE were confirmed by experts (Sinarmas, 2014). In this study the values applied by farmers in the fields are referred to as actual values and also apply to future actual use (step 8).

Step 6. *Determination of current total recommended fertilizer use by crop and province:* Calculation of total recommended fertilizer use was done on basis of the recommended dosage per crop ("fertilizers/area"; see step 4) and the provincial current crop production area (step 2).

Step 7. *Determination of current total actual fertilizer use by crop and province:* The actual determined fertilizer use was calculated on the basis of the recommended values (step 4) with the percentage of actual use (step 5) multiplied with current provincial crop production (step 2).

Step 8. *Determination of recommended and actual future fertilizer use by crop and province:* Future recommended fertilizer use is obtained by applying the provincial crops growth forecasts (step 3) with the recommended actual fertilizer use (step 4). Future actual values were based on combination of future recommended fertilizer and the actual used (step 5). Implicit in this assumption is that the nutrient use efficiency (yield per unit of nutrient input) remains the same. Increased future production can be achieved by increased production per hectare (and thus increased nutrient input per hectare), by increased area under crop production or a combination of these.

Thus, the forecast of actual fertilizer (nutrient and compost) demand (DEM_{Act}) was calculated using the following formula:

$\sum DEM_{act}$ = [Current prod.] x [Growth] x [Fert. req] x [Act/Recom], in which:

$\sum DEM_{act}$: Sum of actual forecasted fertilizer (N, P and K and compost) demand (kt/y) of all identified crops (Table A6.1 in the Appendix Chapter 6);

[Current prod.]: current production in ha/year (Table A6.2 of the Appendix Chapter 6); data were aggregated from provincial data;

[Growth]: growth forecast per crop sector in % (step 3); see Table A6.3 of the Appendix Chapter 6 for future production areas (ha);

[Fert. req]: Recommended fertilizer requirement per ha. For trees, these were converted following a tree density (step 4), using the data presented in Table A6.1 of the Appendix Chapter 6.

[Act/Recom]: Ratio of Actual and Recommended fertilizer requirement (step 5).

The potential P recovery from wastewater that can be recovered in Indonesia from wastewater using struvite precipitation on the side stream of digester supernatant and composting of produced biological sludge is 0.35 kg P/cap/year (cap = capita) (Kerstens et al., 2015). A compost production of 40 kg/cap/year from the treatment of organic solid waste fraction was determined as well (Kerstens et al., 2015). Using a conservative value of a P-content of 0.5% mass based (Veeken et al., 2005), an additional yearly P-recovery of 0.2 kg/cap/year can be expected, which is in line with reported values elsewhere (Strauss, 2003). Thus, the total P-recovery from wastewater and organic solid waste fraction amounts to 0.55 kg P/cap/year. The specific compost production from solid waste and wastewater sludge was 66 kg/cap/year (Kerstens et al., 2015). The potential P-recovery and compost production was based on the expected population of 305 million people by 2035 (BPS, 2013).

6.2.2 Duckweed demand

Duckweed is a valuable protein-rich feedstock for fish production (Journey et al., 1993). In Indonesia, most important freshwater aquaculture comprise monoculture of either tilapia or common carp (Dey et al., 2005). The contribution of carp and tilapia on the total fresh and brackish water aquaculture production amounts to 40% (Dey et al., 2005) and 20% (FAO, 2010), respectively. The combined total of 60% of the freshwater and brackish water fish production was assumed to be potentially fed with duckweed.

Fish production per province (in the period 2005-2011) was obtained through BPS (2014) and categorized by origin. In 2011 produced fish originated from brackish pond (48%), freshwater pond (34%), fish cage (4%), floating fish cage (11%) and rice fields (3%). Cage fish and floating fish can relate to both salt water and fresh/brackish water production. However, no data on the subdivision was available. In this study, it was assumed that 50% of that production consists of fresh/brackish water fish production. Between 2005 and 2009 the total freshwater and brackish water fish production increased yearly by 10%. Between 2009 and 2011 the reported annual growth was 34%. This increase could be the result of actual increased fish production or, alternatively, different or better counting techniques. Because exponential extrapolation of the 2005-2011 growth rates (17%) may result in unrealistic forecasts, future fish production forecast is based on the 2005-2011 (BPS, 2014) linear trend.

An average fish feeding rate of 450 g fresh duckweed/kg fish/d was applied, based on Hassan & Edwards (1992) (660 g/kg fish/d) and El-Shafai et al. (2004) (250 g/kg fish/d). The effect that specific consumption rate tends to decrease when fish grow bigger (Hassan & Edwards, 1992) is excluded (see Appendix Chapter 6, Section 4 and Table A6.4 and A6.5 for a detailed description and provincial breakdown of fish production and duckweed demand). A potential fresh duckweed production of 250 kg/cap/year was assumed following a per capita duckweed production from domestic wastewater of 15 kg dry material/cap/year (Kerstens et al., 2015) and a 6% dry matter content (Hassan & Edwards, 1992).

6.2.3 Plastic production and consumption forecasts and recovery potential

A number of plastics are used in consumer goods: PE (Polyethelene), PP (Polypropylene) and PET (Polyethelene terephthalate) are typically used for plastic bags and bottles, PVC (Polyvinyl chloride) is applied for houseware and toys, and PS (Polystyrene) is often applied for automobile parts and electronics (Syarief, 2006).

In this study the potential recovery of waste plastics to substitute plastic produced in the final steps of the process is analyzed. 2005 data of midstream PE, PP, PVC, PS and PET production were derived from INAplas (*Indonesian Olefin & Plastic Industry Association*) (Syarief, 2006). 2006 and 2012 production data were assessed based on outlook growth numbers of Syarief (2006) and corrected for actual PET production in 2012 (Nurhayat, 2013). These calculated production (5.6%) and consumption (7%) growth rates were used to determine plastic production and consumption till 2025. For the period 2025 until 2035 growth is expected to continue, but a more conservative growth pattern of 5% for both production and growth was used to mitigate a possible overestimation. No literature could be found on long term plastic production and consumption forecasts to verify these long term applied growth rates.

To identify the potential to substitute conventional produced plastic by recovered plastic, the waste plastic recovery was determined. Plastic constitutes 14% of the Indonesian urban solid waste and the majority (80%) can potentially be recycled (Aprilia et al., 2013). In the current study, 75% was applied as a potential recovery to compensate for efficiency loss. Applying a waste production of 2.75 l/cap/d and a densities of 0.25 kg/l (BSN, 1995) a per capita yearly plastic waste production of 35 kg is calculated of which 26 kg/cap/year was assumed recoverable.

6.2.4 Paper production and consumption forecasts and recovery potential

The 2007 paper production was obtained from Kemenperin (2007), whereas the 2013 data and projected 2017 data were obtained from Handoyo (2014). To derive the 2017 planned production capacity, paper production growth until 2024 was defined as 5.5%. Similar to plastic forecasts, production increase between 2025 and 2035 was assumed to slow down and defined as 4%. Indonesian paper consumption grew 7.5% annually between 1992 and 2007, based on Wahyono (2001) and Kemenperin (2007). A more conservative growth of 6% was assumed until the end of 2024 and a 4% growth between 2025 and 2034. No data on long term consumption and production data could be found in literature to verify the long term projection data. The 2011 data on imported waste paper categorized by type and country of import were obtained directly from the MoI (Kemenperin, 2011). Waste paper content of solid waste (12%) was obtained from Aprilia et al. (2013) and a 75% recovery potential (similar to plastic) was applied, resulting in a specific waste paper production of 30 kg/cap/year of which 23 kg/cap/year was assumed recoverable.

6.2.5 Forecast impact analysis

Demand forecasts are subject to uncertainty. Therefore an impact analysis is performed that compares alternative growth forecast of production sectors (staple food, planation, horticulture, aquaculture, plastic, and paper). Three growth forecasts for the year 2035 were compared, being (1) a reduced forecast using an order 2 polynomial growth rate, (2) a linear forecast (order 1 polynomial) and (3) exponential growth rate fitted on the first and last available date. The impact of different forecasts on the fraction of the resource demand that can be recovered was analyzed.

6.3 Results

6.3.1 Fertilizer and compost demand and potential recovery

Figure 6.3 shows the reported and forecasted production quantities for food, plantation and horticultural crops in Indonesia. Numeric values are presented in Table A6.2 (Section 2) of the Appendix Chapter 6.

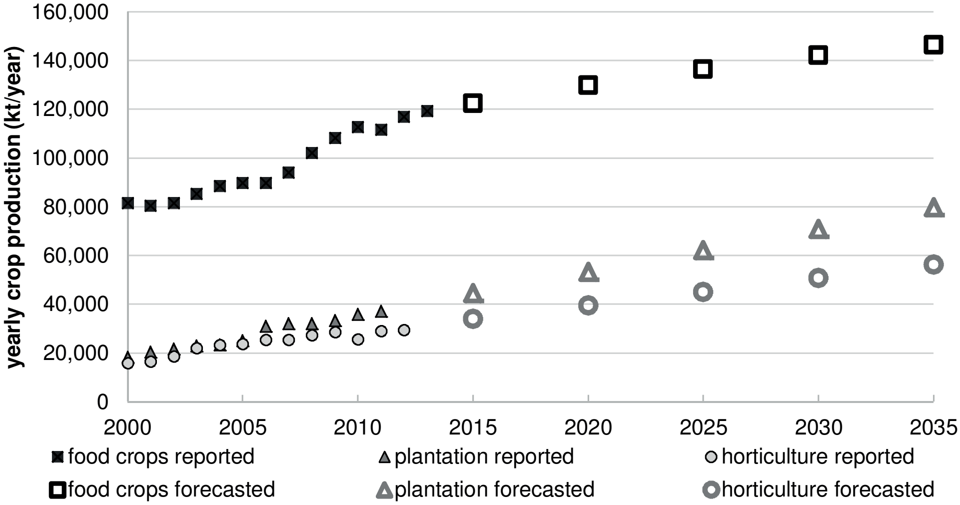

Figure 6.3 Nationwide reported production quantities (solid circles) based on BPS (2014), MoA (2014) and forecasted production (open circles) for food, plantation and horticulture until 2035

Table 6.2 shows the national input of N, P, K and compost based on fertilizer recommendations and on the actual nutrient input per crop and sector. Rice and palm oil are presented separately because of their large contribution to total nutrient demand. An overview of all calculated recommended fertilizer demand per crop is presented in Section 3 (Table A6.3) of the Appendix Chapter 6.

Table 6.2 Indonesian national demand for nitrogen (N), phosphorus (P), potassium (K) and compost as calculated from fertilizer recommendations (Recom) and actual fertilization levels

Sector	Sub sector	Harvested Area (1000 ha/year)[a]	N (kt/year)		P (kt/year)		K (kt/year)		Compost (kt/year)	
			Recom	Actual	Recom	Actual	Recom	Actual	Recom	Actual
Food crops	Total	20,188	2,000	1,200	267	153	749	456	35,619	22,678
	Rice	13,770	1,446	983	162	110	540	367	27,540	18,719
	Others[b]	6,418	554	217	104	43	209	89	8,080	3,959
Plantation	Total	21,884	1,520	938	569	409	1173	774	4,413	2,206
	Palm oil	8,991	382	351	300	276	473	434	0	0
	Others[b]	12,893	1,139	588	269	134	701	341	4,413	2,206
Horticulture		1,908	154	77	46	23	427	213	25,671	12,835
Total		43,980	3,674	2,215	881	586	2,349	1,444	65,703	37,719

[a] Food crops are based on BPS (2014), Plantation and horticulture on MoA (2014)

[b] "others" are non-rice and non-palm oil crops in food and plantation sectors and are presented in Section 3 (Table A6.3) of the Appendix Chapter 6

Figure 6.4 and Figure 6.5 show the 2015, 2025 and 2035 projection of P and compost recommended and actual regional demand per subsector (food, plantation and horticulture). Calculated nationwide data are presented in Table 6.3. Numeric values are presented in the Appendix Chapter 6 (Table A6.6, Section 5).

Table 6.3 Calculated "actual" and recommended P and compost demand forecast in 2015, 2025 and 2035 for Indonesia by sub sector

Resource	Year	"Actual" and recommended value	Total	Food crops	Plantation	Horticulture
P-demand (kt/year)	2015	Actual	675	157	492	26
		Recommended	1,009	273	683	53
	2025	Actual	906	184	686	35
		Recommended	1,344	321	953	70
	2035	Actual	1,130	206	881	43
		Recommended	1,668	357	1,224	87
Compost demand (kt/year)	2015	Actual	40,721	23,269	2,649	14,804
		Recommended	71,448	36,544	5,297	29,607
	2025	Actual	48,359	25,017	3,699	19,644
		Recommended	85,975	39,289	7,397	39,288
	2035	Actual	57,112	27,878	4,749	24,485
		Recommended	102,250	43,783	9,498	48,969

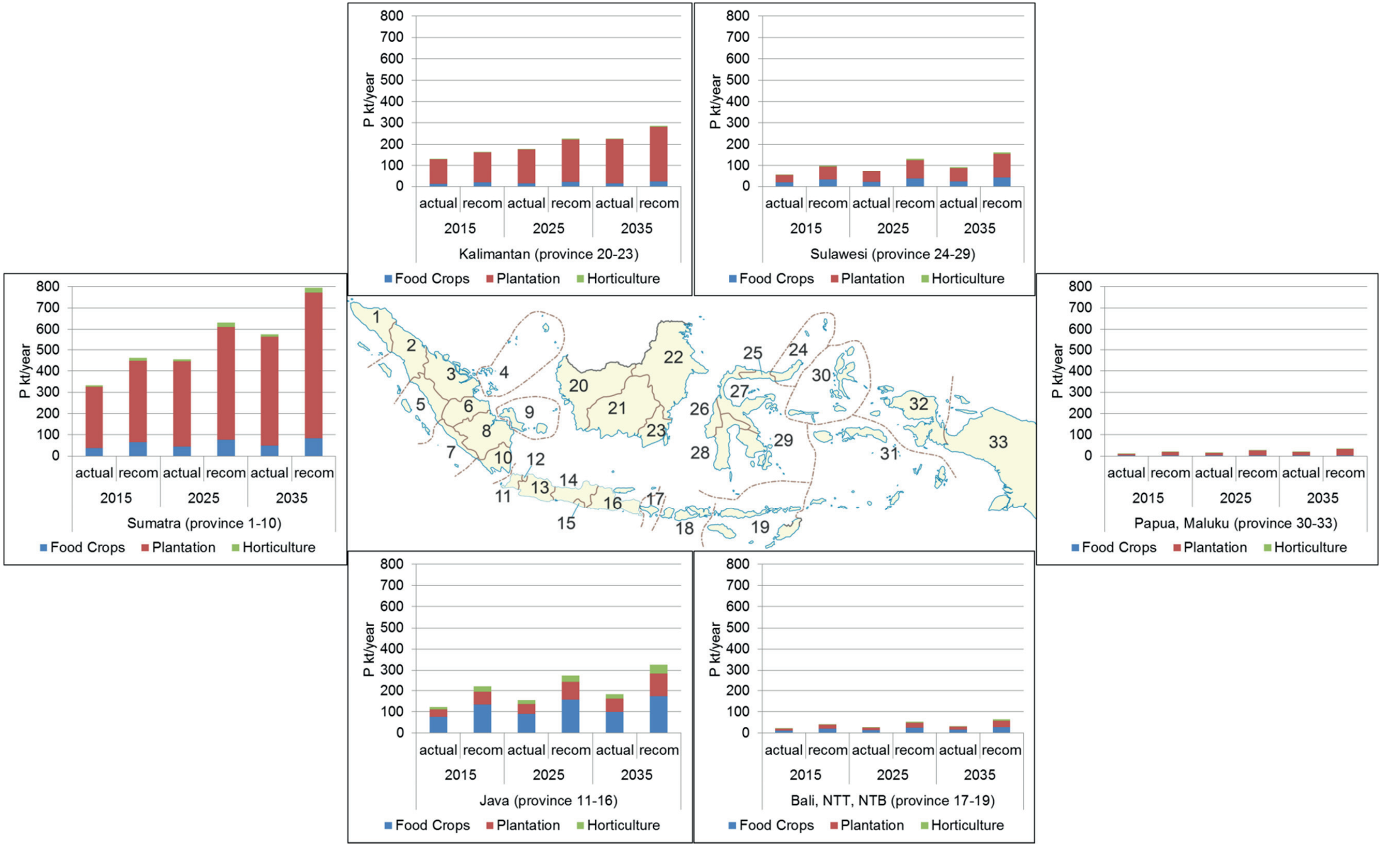

Figure 6.4 Calculated actual and recommended (recom) phosphorus (P) demand for food, plantation and horticulture crops per group of provinces (1-33) for 2015, 2025 and 2035. Sumatra (1-10); Java (11-16); Bali, NTT, NTB (17-19); Kalimantan (20-23); Sulawesi (24-29); Papua and Maluku (30-33). Map from Wikimedia (2014). See also Table A6.6 in Appendix Chapter 6

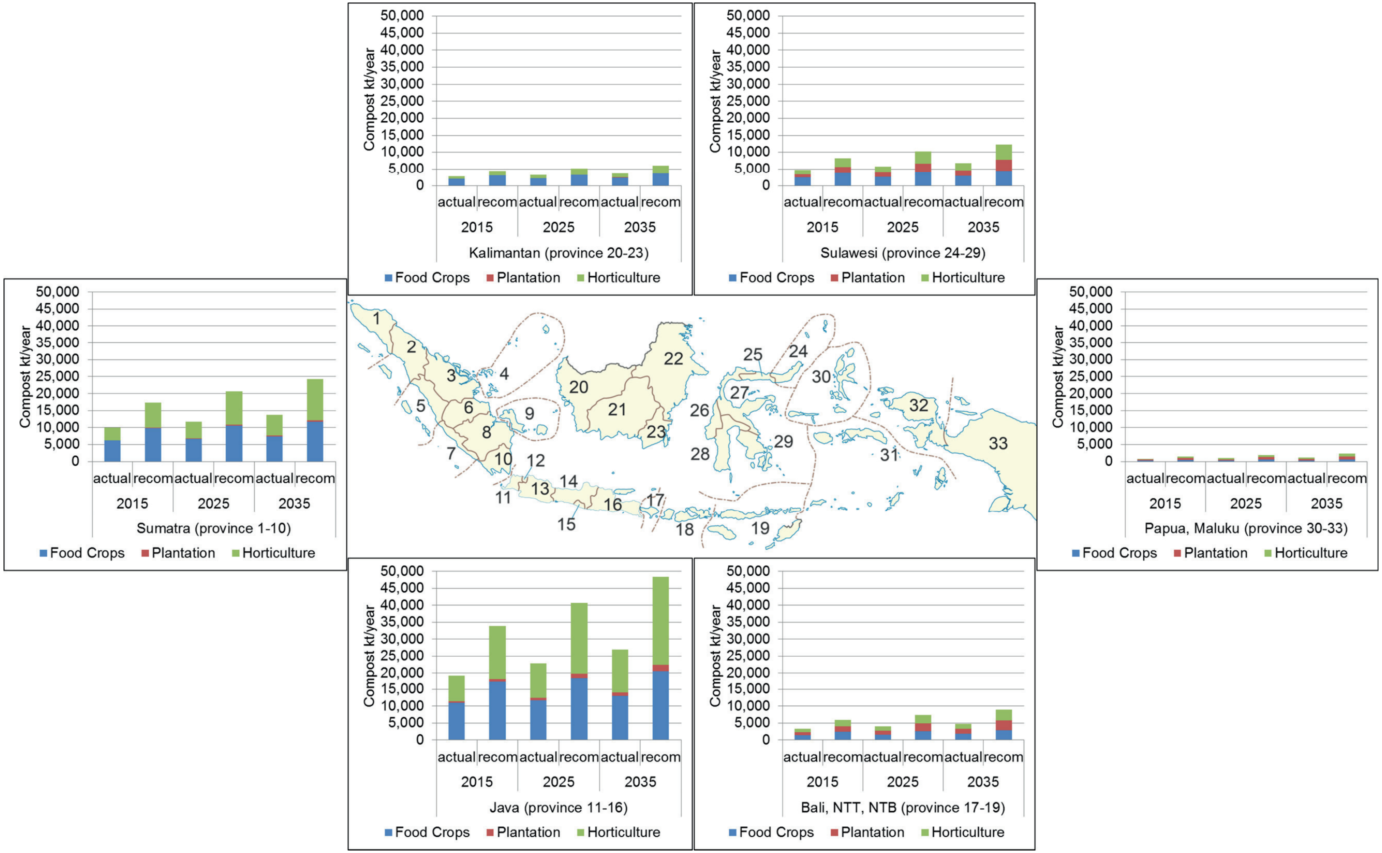

Figure 6.5 Calculated actual and recommended (recom) compost demand for food, plantation and horticulture crops per group of provinces for 2015, 2025 and 2035. Sumatra (1-10); Java (11-16); Bali, NTT, NTB (17-19); Kalimantan (20-23); Sulawesi (24-29); Papua and Maluku (30-33). Map from Wikimedia (2014). See also Table A6.6 in Appendix Chapter 6

In 2035, with an expected population of 305 million people (BPS, 2013), the recoverable P potential will amount to nearly 170 kt/year P. This corresponds with 15% of the actual total P demand and 10% of the recommended P demand in 2035 (Table 6.3). The recoverable compost by 2035 amounts to 20,130 kt/year, which amounts to nearly 20% and 35% of the recommended and actual demand, respectively.

6.3.2 Duckweed demand and recovery potential

Figure 6.6 shows the potential calculated fresh duckweed demand development per region (see Figure 6.4 for the regions) in the period 2015-2035.

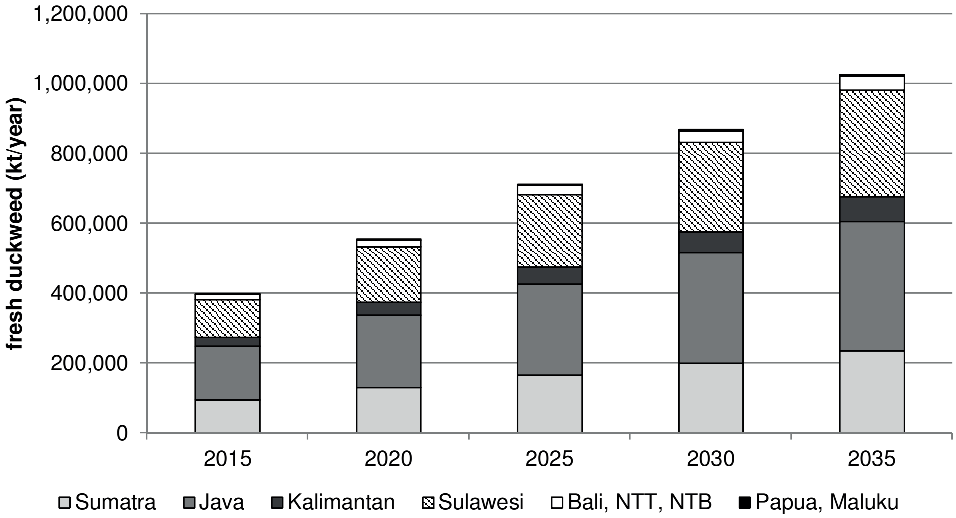

Figure 6.6 Calculated fresh duckweed demand development 2015-2034 per region in Indonesia

Java accounts for nearly 40% potential duckweed demand while half of all Java fishery is located in West Java (BPS, 2014). Sulawesi accounts for about 30% of all potential duckweed demand. For a 305 million people population the maximum fresh duckweed production by 2035 corresponds with 75,000 kt/year. This equals 7% of the calculated (potential) demand.

6.3.3 Plastic production and consumption forecasts and recovery potential

Figure 6.7 shows the reported Indonesian production and consumption of plastics (2005 and 2012) and the extrapolated future developments.

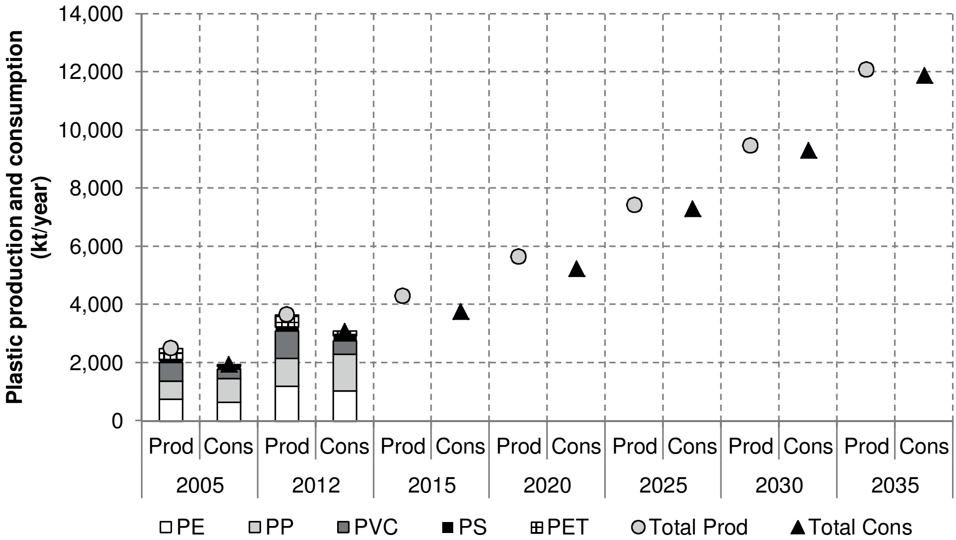

Figure 6.7 Indonesian reported production (Prod) and consumption (Cons) data (Syarief, 2006; Nurhayat, 2013) and extrapolated future data. For the years 2005 and 2012 subcategories (PE, PP, PVC, PS and PET) are provided, while forecasts only show total quantities

With a population of 305 million people (BPS, 2013) approximately 8,000 kt plastics can be recovered annually in 2035 (Table 6.4). This corresponds with 66 and 68% of the forecasted production and consumption respectively.

Table 6.4 Estimated plastic consumption, production (kt/year) and recovery (%) in 2035 in Indonesia

Plastic production	Plastic consumption	Calculated recoverable plastic	Recovered as part of production	Recovered as part of consumption
12,086	11,892	8,036	66%	68%

6.3.4 Paper production and consumption forecasts and recovery potential

Figure 6.8 shows the reported and forecasted Indonesian paper production and consumption. The total demand for waste paper in 2011 amounted to 5 million t/year (APKI, 2012).

The total amount of imported waste paper in 2011 (Figure 6.9) was 2,450 kt and equals about half of the total waste paper demand (APKI, 2012). More than 60% of the waste paper is supplied as Old Corrugated Cardboard and the main source of imported waste paper is Singapore, accounting for almost 20% of the total (Figure 6.9).

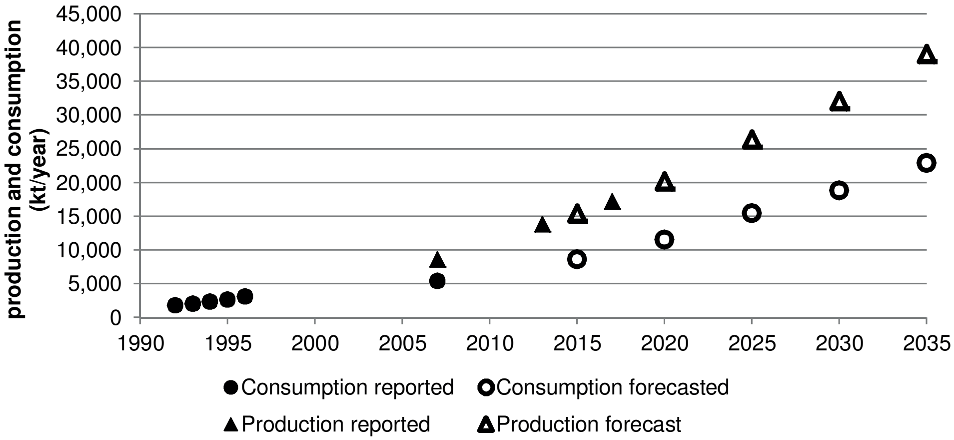

Figure 6.8 Reported (solid rounds and triangles) and forecasted Indonesian paper production and consumption (Wahyono, 2001; Kemenperin, 2007; Handoyo, 2014)

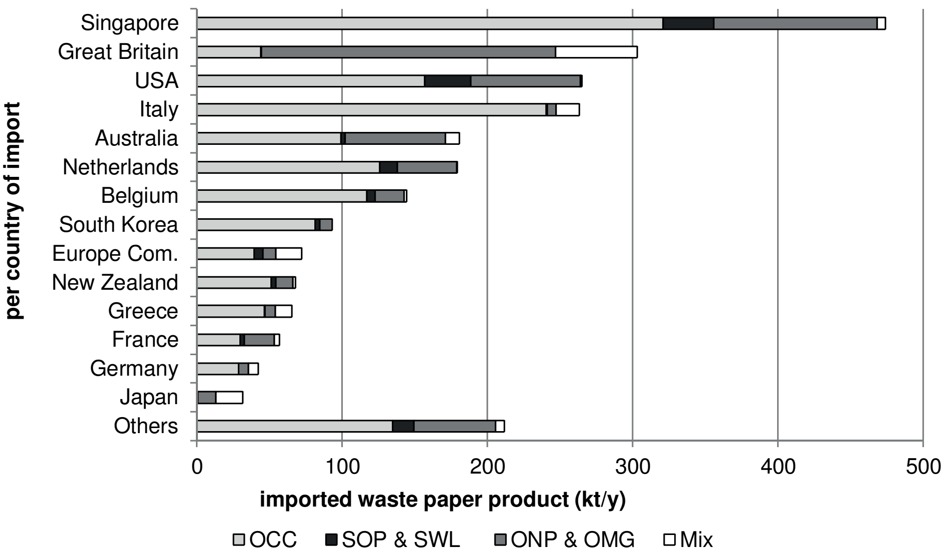

Figure 6.9 Imported to Indonesia waste paper per type and origin in 2011 (Kemenperin, 2011): Old Corrugated Cardboard (OCC), Sorted Office Paper (SOP), Sorted White Ledger (SWL), Old Newspaper (ONP), Old Magazine (OMG) and Mixed Paper (Mix). “Others” comprise 35 countries that together contribute 10% of the total

In 2013 the total paper production amounted to 13.9 million t/year (Handoyo, 2014) of which an estimated 7.7 million t/year was consumed locally. The remaining 45% was exported, which is in line with the percentage reported by the Ministry of Industry (Kemenperin, 2007). Based on extrapolation of the 2011 data, 3 million t/year of waste paper is expected to be imported to Indonesia by 2015 (Table 6.5). With an estimated total demand of 5.8 million t/year of waste paper, APKI (2012), local demand for waste paper exceeds 3.1 million t/year by 2015 and potentially 6.9 million t/year by 2035. This corresponds with 18% and 31% of the 2035 forecasted production and consumption, respectively. It is in line with the calculated amount of 6.9 million t/year demand for waste paper by 2035 (Table 6.5).

Table 6.5 Estimated paper consumption, production, import and recovery in Indonesia for 2015 and 2035

Parameters	Unit	2015	2035
Paper production	kt/year	15,471	39,118
Paper consumption	kt/year	8,641	22,907
Total waste paper demand	kt/year	5,871	14,845
Waste paper imported	kt/year	3,033	7,669
Local waste paper demand	kt/year	2,838	7,176
Recovery potential	kt/year	3,052	6,888
Recovery potential as part of production	%	20%	18%
Recovery potential as part consumption	%	35%	31%

6.3.5 Forecast impact analysis

In Table 6.6 the 2035 forecasted production quantities used in this study are compared with (1) an order 2 polynomial growth rate, (2) a linear forecast and (3) exponential growth rate.

Table 6.6 Impact analysis on 2035 production and consumption quantities (kt/year) per sectors using an order 2 polynomial growth, a linear growth and exponential growth

	Forecasts			
Sector	Used in this study[a]	Order 2 Polynomial growth	Linear growth	Exponential growth
Food crops production	146,000	168,000	191,000	226,000
Plantation production	80,000	61,000	80,000	172,000
Horticulture production	56,000	47,000	56,000	97,000
Fish (tilapia and carp) production	6,000	4,000	6,000	83,000
Plastic consumption	12,000	5,000	7,000	14,000
Plastic production	12,000	6,000	7,000	13,000
Paper consumption	23,000	9,000	12,000	41,000
Paper production	39,000	25,000	33,000	65,000

[a] Growth forecast used in this study are explained in materials and methods

6.4 Discussion

To identify the potential to close material cycles in the Indonesian context, we first discuss the boundary conditions for our evaluation, the actual demand and recovery potential and forecasted developments in their regards. Second, we identify policy-related issues that should be considered to facilitate a transition from a *business as usual* scenario to a circular economy. Third, we list additional resources (e.g. nitrogen, algae) that can be recovered from wastewater and solid waste. Fourth, we evaluate the applicability to use the presented methodology for other countries.

6.4.1 Potential resource demand and recovery analysis

To verify the calculated demand and recovery potentials, we discuss the boundary conditions and results for the resource that were described in Sections 2 and 3. We then identify how the described outcomes are subject to uncertainties in growth forecasts and potential different uses of resources.

6.4.1.1 Verification of resource demand and recovery potential

Phosphorus:

By 2035 an estimated actual demand of 1,130 kt P per year is calculated of which 78% will be used by plantations (Table 6.3). Table 6.2 showed that nearly 50% of the calculated current actual P demand is required for the palm oil sector. By 2035 over 70% of the total is expectedly demanded by plantations (Table 6.3, Figure 6.4). The majority of palm oil plantations are located in Sumatra and Kalimantan (MoA, 2014) and more than 60% of all P is expectedly needed in these areas.

The presented growth of plantation crop production (Figure 6.3) is smaller than forecasted by Bambang (2011), but still shows a 100% increase from 2010 to 2035. In comparison for Malaysia, a 50% increase of domestic palm oil production is projected for 2035 compared to 2009 (Gan & Li, 2014). A decreased growth rate in future for Indonesia compared to past growth rates as used by Bambang (2011) seems plausible in the light of restricted expansion of new palm oil plantation areas (Gan & Li, 2014).

The calculated actual N, P and K demand were compared with national statistics of FAO (2014a) on fertilizer use (FAOSTAT). There is a considerable discrepancy between the calculated actual N, P and K values in Table 6.2 and the data for 2011 from the FAOSTAT database. FAOSTAT shows a demand of 2,923 t N/year, 252 t P/year and 854 t K/year, while the calculated (actual) values were 2,215 t N/year, 586 t P/year and 1,444 t K/year (Table 6.2). The reported N use by FAO is higher than the calculated actual value, whereas for P and K these are lower than calculated. Several factors are identified that may explain this discrepancy (see also Appendix Chapter 6, Section 6 for a detailed analysis):

1. *Possible incomplete data used by FAOSTAT*: Analysis of obtained datasheet suggests that not all private estates, and their associated P-consumption by palm oil plantations (Table 6.2) are included (IFPA, 2013; FAO, 2014a). A total P-demand of 310 t/year P compared with 252

t/year P (FAOSTAT) is calculated in case P-consumptions by palm oil plantations (276 t/year P Table 6.2) are excluded.

2. *Difference in used recommended and actual fertilizer rates in calculations*. Discrepancy in calculated fertilizer demands and FAOSTATS values may further be related to applied fertilizer rates (Padjajaran University, 2013) that differ from other available guidelines (FAO, 2005; Bambang, 2011); Table 6.7 shows the recommended rice fertilizer rates in this study compared to reference rates. Appendix Chapter 6, Section 6 provides an additional analysis on fertilizer use for palm oil plantations.

Table 6.7 Recommended rice fertilizer rates (kg/ha/y) in this study compared to reference rates

Source for fertilizer rate	N	P	K
This study (Padjajaran University, 2013)	105	12	39
FAO (2005)	60-100	15-25	4-35
Bambang (2011)	70	20	14

3. *Farmer's interpretation on fertilizer guidelines*. Prices for KCl and SP-36 (superphosphate, 36% P_2O_5) in Indonesia are higher than urea following the ministerial guideline (Kariyasa, 2005; MoA, 2012c). Consequently, farmers reportedly use KCl and SP36 as complementary fertilizers, whereas urea is regarded as the main fertilizer in farming (Kariyasa, 2005). This may result in a lower K and P use (as reported in FAOSTAT) compared to recommended values (and calculated actual values).
4. *Use of organic fertilizer as a source of N, P and K*. FAOSTAT only registers produced inorganic fertilizers, whereas organic fertilizers are not included. Organic fertilizers also contain N, P and K. Chicken manure is a popular organic fertilizer in Indonesia (Buresh et al., 2010). In addition, sheep manure production is reported as important as an output as meat production (Tanner et al., 2001). The total N and P to be expected from animal manure is approximately 400 and 150 kt/year N and P (see Appendix Chapter 6, Section 7 for calculation), but it cannot fully explain the discrepancy.
5. *Use of phosphate* rock. Direct use of untreated phosphate rock as an alternative to commercial mineral fertilizers as SP or TSP (trisodium phosphate) is applied in some regions in Indonesia (Yusdar et al., 2007). Phosphate rock is not accounted for in the FAO statistics (FAO, 2014b) and application would therefore not be included.

The P-recovery potential for Indonesia is lower than reported values elsewhere. Mihelcic et al. (2011) determined that the phosphorus present in urine and feces corresponds with 22% of the global P demand, whereas De Graaff et al. (2011) reported that recovery of P from wastewater as struvite equals 10% of the current artificial phosphorus fertilizer production in the world (excluding P-recovery from solid waste). Lower percentages for Indonesia compared to reported literature are attributed to high assumed P fertilizer demand for plantations (Table 6.2)

Compost:

The forecasted recommended horticultural compost demand by 2035 ranges from 24 (actual) to nearly 50 million t/year (Table 6.3). This corresponds with 42% and nearly 50% of the total calculated compost demand. The total amount of compost demanded in Indonesia may grow from the recommended quantity of 65,000 kt/year (Table 6.2) to more than 100,000 kt/ year in 2035 (Table 6.3).

Compost as an organic fertilizer is considered an important resource for horticulture (Aye & Widjaya, 2006; Rogger et al., 2011) and may increase disease suppressiveness of crops (Veeken et al., 2005). In 2001 the Indonesian potential compost demand was estimated as 11 million t/year (Aye & Widjaya, 2006), which is only 20% of the actual and 10% of the recommended 2035 determined demand. Compost is further recognized as a beneficial amendment in soil improvement (Moeskops et al., 2010) in, for example, urban landscaping (Sloan et al., 2012). The latter option was not included in the current analysis due to lack of local information on landscape activities.

Duckweed:

No information on duckweed demand for aquaculture could be found in literature to compare our data with. However, utilizing duckweed in its fresh, green state as a fish feed minimizes handling and processing costs and is safe to use, while nutritional requirements of fish are met completely (Journey et al., 1993; Islam et al., 2004). The protein content of dry duckweed is comparable to or higher than that of other common fish feed ingredients such as fish meal, soy bean, water hyacinths or alfalfa (Journey et al., 1993; Cheng & Stomp, 2009).

Plastic:

The specific plastic recovery value of 35 kg/cap/year fits well with reported values in other Asian countries, such as Malaysia (26 kg), Thailand (30 kg), Singapore (45 kg) or Japan (80 kg), but is higher than reported 2006 Indonesian values of 9.5 kg/year (Syarief, 2006). Pasang et al. (2007) and Chaerul et al. (2013) reported waste plastic production approaching 35 kg/cap/year for Jakarta and high income households in Bandung. However, the average reported plastic waste production in Bandung was below 10 kg/cap/year (Chaerul et al., 2013). The majority (73%) of plastic waste concerns plastic packaging waste (Chaerul et al., 2013). In view of the increasing plastic production (Syarief, 2006; GBGIndonesia, 2013) and an average increase of 8% in plastic shopping bags (Cornelia et al., 2013) values approaching 50 kg/cap/year could be expected by 2035 from the current 10 kg/cap/year.

Paper:

Paper production has shown a continuous increase since the industry experienced a spectacular expansion in the 1990s in which output increased nine fold (Jonker et al., 2006). Even during the

Asian crisis, total sector production capacity continued to grow as the collapse of domestic demand was compensated by an increasing export orientation (Jonker et al., 2006). The presented paper recovery values (23 kg/cap/year) relate to household waste generated only and excludes paper recovery and recycling from waste office paper. In 2008, the amount of waste collected from offices was less than 10% of the total (KNLH, 2008), but sectorial waste generation data from neighboring countries show that waste from offices and institutions can amount to 30% of the total (Saeed et al., 2009).

6.4.1.2 Sensitivity of forecasts

The presented forecast demand analysis is subject to uncertainties. These uncertainties relate to variations in growth forecasts (Table 6.6), but also in different future uses of the studied resources.

Our analysis used long-term (typically 10 years) historical data to predict the future demand. Following an impact analysis using different growth rates, minimum and maximum demand values could be identified (Table 6.6). The lower used plantation forecast compared to the exponential growth rate was based on changing policies in neighbouring countries that may impact Indonesia as well (Gan & Li, 2014).The large difference for fish production in Table 6.6 between the exponential growth and order 2 polynomial growth or linear growth was attributed to the reported huge increase between 2009 and 2011. Paper and plastic consumption and production show the biggest variations when comparing the minimum values with used values (Table 6.6). Bandara et al. (2007) showed that with increasing income the paper waste production increases. Therefore, applied growth rates for the paper and plastic sector seem justified in view of increasing economic growth per capita (ADB, 2012) and increases of waste plastic (Cornelia et al., 2013).

Variation in future growth rates affects the potential recovery fraction in relation to the demand. The minimum potential recovery is obtained with highest growth forecast and vice versa. For example, the calculated maximum potential recovery of compost in relation to the (actual) demand in 2035 in Indonesia can be as high as 39% and as low as 21%.

Table 6.8 shows the impact of maximum and minimum applied sector growth rates on the potentially recoverable fraction of the demand (see also Appendix Chapter 6, Section 8).

Table 6.8 Impact of maximum (exponential growth rates) and minimum (order 2 polynomial) growth rates on % potential recoverable resources in 2035 for Indonesia in relation to the Recommended (Recom) and Actual demand and Consumption (Cons) and Production (Prod)

Resource	P		Compost		Duckweed	Plastic		Paper	
	Recom	Actual	Recom	Actual		Cons	Prod	Cons	Prod
Baseline [a]	10%	15%	20%	35%	7%	68%	66%	30%	18%
Minimum [b]	5%	7%	12%	21%	1%	58%	61%	17%	11%
Maximum [c]	12%	19%	22%	39%	11%	162%	133%	77%	27%

[a] Baseline uses the growth forecast applied in this study; [b] Minimum potential recovery is obtained with highest growth forecast; [c] Maximum potential recovery is obtained with lowest growth forecast

In addition to uncertainties in growth forecast, future demand may be affected by different uses of resources. In this paper we assumed that recovered resources will be applied for indicated purposes. However, duckweed, as an example, is also considered as a replacement for animal-derived proteins to enter the European market (van der Spiegel et al., 2013). In addition, duckweed may be used as a source for bio-fuel production and is regarded a promising alternative for bioenergy production (Cheng & Stomp, 2009; Verma & Suthar, 2015). These factors may be an additional driver for duckweed demand. A second example of how new developments may affect forecast is the introduction of environmental friendly plastic bags, made from renewable raw materials, such as starch from cassava, corn or others (Cornelia et al., 2013). These may become an alternative to conventionally produced or recycled plastics. The impact of such developments is not further quantified in the current study, but will eventually affect demand for resources, such as duckweed and plastics.

6.4.2 Resource recovery as an option to close material cycles

Insight in demand and supply of recoverable resources may be an incentive for policy makers to apply a certain type of wastewater or solid waste technology to assure long-term availability and sustainability of scarce (locally available) resources. Long-term availability of worldwide phosphorus (Cordell et al., 2011), consumer goods productions (Raitzer, 2010), and, to a lesser extent, organic soil in Java (Minasny et al., 2011) are under pressure. Ignoring valuable resources in wastewater and solid waste and continue current practices may jeopardize the security of future food supply. In addition, resource recovery in the Indonesian context can be financially more attractive than the application of conventional (non-resource recovery) systems (Kerstens et al., 2015). Especially in Java, the potential to recover resources is considerable, because resource recovery technologies are feasible and nearly 60% of the Indonesian population lives in Java (Kerstens et al., 2015). To shift from a *business as usual* scenario to a circular economy, a number of issues should be considered.

First, the safe use and social acceptability of recovered resources should be assured. P and compost demand for agriculture are higher than the recovery potentials from wastewater and solid waste (Table 6.8). Even when applying low sector growth rates, the maximum recovery of P and compost are, respectively 19% and 39% (Table 6.8). When demand exceeds supply, selective marketing of recovered resources or products, which pose no direct threat to humans, should be strived for. Potential markets for processed fecal sludge are palm trees, rubber plantations as well as fruit trees (WHO, 2006b; Almy, 2008). In addition to health aspects, social acceptability of sludge products must be considered. Starkl et al. (2010) describe reluctance in use of by-products from human feces, despite (potential) financial benefits. Hence, policy makers should consider (1) the origin of recovered resources (e.g. identify the need for source separation), (2) level of hygienization of sanitation by-products, and (3) the perception of envisaged users (Koné et al., 2007; Snyman & Vorster, 2011; Raschid-Sally, 2013).
Second, a circular economy is facilitated when distances between resource demand and supply locations are short (Agudelo-Vera et al., 2011). The P-demand is highest in Sumatra, followed by Java and Kalimantan. However, only 27% of all 305 million Indonesian people in 2035 will live in Sumatra and Kalimantan, whereas these two islands require over 60% of the total P (Figure 4). Phosphorus can be recovered for example as struvite or as a nutrient in compost from sewage sludge or organic solid waste in (medium) centralized wastewater or municipal solid waste systems (Kerstens et al., 2015). However, the application of (medium) centralized (off-site) systems is only financially feasible in densely populated urban areas (Kerstens et al., 2015). Urbanized areas can be typically found in Java, but far less in Sumatra and Kalimantan, where the number of people living in urban areas will be less than 55% in 2035 (BPS, 2013). The majority of people in Sumatra and Kalimantan will live in rural areas where septic tanks and pit latrines are the prevailing sanitation service. Van Voorthuizen et al. (2008) reported that 95% of the P is lost to the effluent when applying septic tanks. As a result the P-demand and P-supply locations do not match. Compost production is possible from wastewater sludge and solid waste and can be applied on a central and decentral level (Kerstens et al., 2015). The wider application possibilities of composting plants may result in a better compost marketing potential compared to P marketing. The production of proteins from wastewater using duckweed ponds is a feasible option for Indonesia, but requires off-site systems and a large footprint (Kerstens et al., 2015). In view of increasing land prices (Navastara & Navitas, 2012) potential benefits of duckweed production should outweigh the additional land costs. Application should be considered for locations adjacent to urban areas with surrounding aquaculture activities, such as West Java and Sumatra.
Third, policy makers should consider the different stakeholders in the sanitation planning process. Recycling of waste plastic and paper products has been predominantly established in Indonesia by the informal sector (Chaerul et al. 2013; GBGIndonesia 2013; APKI 2012). To safeguard this apparent financially sustainable system, experiences and organization structures and the role of

informal sector should be considered (Sasaki & Araki, 2013). The waste plastic and paper processing industries should also be involved in the planning process. In 2009 there were about 60 registered manufacturers processing recycled materials in Java (Febrina, 2009). The fastest growing waste plastic recycling industries are in East Java where production is reported to fulfill 70% of the plastic pellets needs of East Java (Arifianto & Fajar, 2012). The quality of recovered raw materials (e.g. plastics) largely determine their potential to become a replacement of virgin plastics (Lazarevic et al., 2010) and may require specific policy interventions to stimulate plastic recovery (e.g. introduction of deposits on plastic bottles). In addition, Van Beukering (2001) showed that developing countries, such as Indonesia, specialized in the utilization of waste paper. Policy makers can support international trade flow discussions to stimulate the use of waste paper rather than wood pulp to develop a more sustainable Indonesian pulp and paper sector (Raitzer, 2010). Currently, a national 20% recovery target of urban waste is proposed for Indonesia (Bappenas, 2014), resulting in a lower plastics and paper recovery potential than the calculated recoverable amount. Therefore, in the formulation of recovery targets, the current and planned presence of locally based recycling facilities, their local and international stakeholders and their needs should be included.

6.4.3 Additional recoverable resources and their demand

Besides the described resources, there are a number of other resources or products that can be recovered from wastewater and solid waste for which markets exist. Nitrogen is another essential plant nutrient that is present in wastewater. Processes for the conventional production of N fertilizer are based on the energy intensive fixation of atmospheric nitrogen in the Haber-Bosch process. Compared to the energy requirement for the Haber-Bosch process, N-recovery from wastewater can be energetically favorable (Tervahauta et al., 2013). However, N-recovery requires high concentrated flows (e.g. separately collected urine or digester supernatants) and relative complex technologies (e.g. stripping, ion exchange) (Maurer et al., 2006). These technologies are not found feasible in the current Indonesian context and therefore excluded from the current study. Wastewater treatment using duckweed ponds on the other hand is an alternative and low cost option of recovering nitrogen in the form of proteins (Bal Krishna & Polprasert, 2008).

Besides duckweed, algae can be used to recover nutrients from wastewater. The use of algae ponds as a treatment has been widely practiced in warm climates (Van der Steen et al., 1998; El-Shafai et al., 2007). Compared to duckweed, the recovery of the algal biomass is complicated and only possible by using cost- and knowledge intensive techniques such as flocculation-flotation using chemicals and centrifugation or combinations of these (Oron et al., 1987; El-Shafai et al., 2007).

Besides paper and plastic recycling, energy recovery through incineration of paper and plastics may have potential benefits (Chen & Chen, 2013). However, incineration is not yet found feasible in Indonesia (Aprilia et al., 2012) and was therefore not further analyzed in this study.

6.4.4 Applicability of used methodology in other countries

The methodologies used in the paper are based on publicly available data from the statistical bureau of Indonesia and FAO, supplemented with country specific fertilizer practises and interviews with local stakeholders (ministries, associations and universities). This makes the methodology for the spatial resource demand for crop nutrients (see framework in Figure 6.2), proteins and recoverable plastics and papers applicable for other countries as well. The collected data on fertilizer demand per crop area in the online supplementary information section 1 (required in step 4 in Figure 6.2) can directly be used as indicative values for other tropical countries. For non-tropical countries these values can be determined using FAO databases. Presented methodology and data (Kerstens et al., 2015) for specific annual per capita recovery potentials (P, compost, duckweed and plastics/papers) from wastewater and municipal solid waste allows for comparing the recovery potential in relation to their demand.

Changing populations and distribution and the associated impact on food systems and finite resources make the determination of future demand for resources and their local distribution an important element in resource management. While many countries struggle in the implementation of wastewater and solid waste facilities, various technologies are available to recover resources from wastewater and solid waste. When planning wastewater and solid waste infrastructures, the developed methodology can thus contribute to a circular resource management while assuring a favorable financial perspective.

6.5 Conclusions

The methodology presented in this paper enables a comparison between resource demands from agriculture, aquaculture and the consumer goods sector on the one hand and resource recovery potential from wastewater and solid waste on the other. Applying our methodology for the case of Indonesia, we can conclude that the current poor state of the Indonesian wastewater and solid waste sector offers opportunities for the direct introduction of a circular resource management system. Such opportunities manifest in areas where high resource recovery potentials from municipal wastewater and solid waste match their demand, such as the urban areas on Java.

This study demonstrated that by 2035 significant fractions of the Indonesian demand for phosphorus (15%), compost (35%), duckweed (7%), plastic (66%), and paper (18%) can be satisfied by resource recovery from wastewater and solid waste. Since resource demand exceeds potential recovery, selective marketing of resources, focusing on their safe use, should be considered.

The potential demand for recovered P and compost is highest in Sumatra and Java. Large potential duckweed demands are anticipated in Sulawesi. However, there is a geographical discrepancy between the potential recovery and demand location of, especially, P and compost. In the planning and selection of wastewater and solid waste facilities the location of potential markets (agriculture and aquaculture) should be considered.

Recovery of waste paper and plastic can significantly satisfy the already established demand for these products and substitute products manufactured in conventional processes using wood or non-renewable fossil fuels.

The methodology and data may be applied in other countries as well to assess the potential of recovery of resources from wastewater and solid waste to satisfy resource demand.

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Appendix Chapter 6

Section 1 Overview of studies crops, fertilizer requirements and tree densities

Table A6.1 Overview of studied crops, recommended fertilizer requirements and tree densities, based on Padjajaran University (2013), FAO (2005), MoA (2012a), MoA (2012b), MoA (2014) and Weiss (2002)

Type	Number	Crop name	kg fertilizer/ha				Fertilizer/tree				Tree density
			N	P	K	Organic	N	P	K	Organic	trees/ha
(Staple) food Crops	1	Paddy	105	12	39	2,000					
	2	Corn	117	16	26						
	3	Soybean	12	24	39						
	4	Peanut	0	16	26	3,000					
	5	Greenpeal	23	10	26						
	6	Cassava	70	16	52	5,000					
	7	Sweet Potato	82	16	52	5,000					
Plantation	8	Sugar Palm					0.45	0.45	0.45		204
	9	Cloves					0.16	0.08	0.12	12.50	200
	10	Cashew Nut					0.16	0.03	0.18	50.00	100
	11	Cacao					0.03	0.01	0.03		1,111
	12	Rubber					0.15	0.04	0.14		550
	13	Cinnamon	2	1	1						
	14	Coconut	100	22	41						
	15	Palm Oil					0.32	0.25	0.39		134
	16	Coffee					0.09	0.02	0.07		1,300
	17	Pepper					0.06	0.03	0.03		2,000
	18	Nutmeg	8	2	4	2,500					
	19	Areca Nut	70	47	72						
	20	Sugar Cane	308	32	90						
	21	Tea	150	44	41						
	22	Tobacco	58	21	56						
Horticulture	23	Shallot	70	39	131	15,000					
	24	Onion	70	30	105	15,000					
	25	Potato	187	63	131	30,000					
	26	Cabbage	117	31	105	15,000					
	27	Tomato	70	20	26	20,000					
	28	Cayenne Pepper	70	39	78	20,000					
	29	Chive	560	0	0	12,500					
	30	Cauliflower	107	63	94	17,500					
	31	Chinese Cabbage	233	40	126	17,500					
	32	Carrots	70	20	26	15,000					
	33	Radish	93	80	0	10,000					
	34	Red Bean	23	10	26	0.0					

	35	Long Bean	23	40	65	17,500					
	36	Chili	117	30	39	15,000					
	37	Paprika	47	40	26	12,500					
	38	Terung	140	121	94	0.0					
	39	Bean	28	24	52	15,000					
	40	Cucumber	56	50	47	20,000					
	41	Kangkung	70	0	0	5,000					
	42	Spinach	47	20	42	10,000					
	43	Melon (Tree)					0.00	0.00	0.00	1.00	12,500
	44	Watermelon (Tree)					0.00	0.00	0.79	1.00	12,500
	45	Cantaloupe (tree)					0.00	0.00	0.00	1.00	18,000
	46	Melinjo (Tree)					0.13	0.03	0.13	40.00	192
	47	Petai (Tree)					0.13	0.03	0.13	40.00	63
	48	Jengkol (Tree)					0.13	0.03	0.13	40.00	72
	49	Avocado (Tree)					0.78	0.00	0.00	10.00	100
	50	Starfruit (Tree)					0.06	0.10	0.20	25.00	300
	51	Duku/ Langsat (Tree)					0.01	0.01	0.01	120.00	100
	52	Durian (Tree)					0.08	0.10	0.08	40.00	100
	53	Guava (Tree)					0.16	0.03	0.18	50.00	300
	54	Pink Water (Tree)					0.05	0.01	0.03	60.00	100
	55	Orange (Tree)					0.10	0.04	0.10	20.00	374
	56	Mangga (Tree)					0.08	0.03	0.08	20.00	100
	57	Mangoesteen (tree)					0.02	0.00	0.01	20.00	100
	58	Jackfruit (Tree)					0.16	0.03	0.18	35.00	100
	59	Pineaple (Tree)					0.00	0.00	0.00	0.50	24,998
	60	Papaya (Tree)					0.09	0.03	0.07	40.00	1,000
	61	Banana (Tree)					0.16	0.03	0.18	50.00	1,000
	62	Rambutan (Tree)					0.09	0.03	0.13	32.50	100
	63	Salak (Tree)					0.05	0.00	0.00	17.50	1,581
	64	Sapodilla (tree)					0.18	0.00	0.00	2.00	79
	65	Passionfruit (tree)					0.00	0.00	0.00	15.00	886
	66	Soursop (Tree)					0.12	0.06	0.08	17.50	286
	67	Apple (Tree)					0.05	0.00	0.00	40.00	615
	68	Grape (Tree)					0.35	0.16	0.26	20.00	982

Another 7 plantation crops (Tamarind, cotton, kapuk, Vanilla, Fiber Sack/ Hemp, Lemongrass and Siwalan) and 3 horticultural crops (squash, strawberry and breadfruit) were studied of which only area data was available, but no fertilizer requirements. The seven "missing" plantation crops amounted to 1% of the total plantation area; the missing horticultural area accounted for 1% of the total horticultural area.

Section 2 Reported and forecasted production quantities for food, plantation and horticulture (2000-2035)

Table A6.2 Nationwide reported production quantities (kt/y) based on BPS (2014), MoA (2014) and forecasted production (italic font starting from 2015) for food, plantation and horticulture until 2035

Year	Food crops	Plantation	Horticulture
2000	81,536	18,307	15,835
2001	80,449	20,257	16,601
2002	81,508	21,586	18,689
2003	85,332	22,927	21,961
2004	88,512	23,408	23,262
2005	89,819	24,975	23,732
2006	89,807	31,024	25,527
2007	94,024	31,906	25,503
2008	102,126	32,020	27,413
2009	108,193	33,244	28,702
2010	112,744	35,757	25,713
2011	111,524	37,023	29,008
2012	116,944	-	29,428
2013	119,163	-	-
2015	*122,410*	*44,444*	*33,940*
2020	*129,887*	*53,254*	*39,489*
2025	*136,482*	*62,065*	*45,038*
2030	*142,029*	*70,876*	*50,587*
2035	*146,460*	*79,687*	*56,136*

Section 3 Recommended fertilizer per studied crops

Table A6.3 Calculated recommended nutrient demand for Indonesia food (base year 2013), plantation (base year 2011), and horticulture (base year 2012)

type	number	Crop name	N-total kt/y	P-total kt/y	K-total kt/y	Compost kt/y
(Staple) food Crops	1	Paddy	1,445.8	162.3	540.3	27,539.8
	2	Corn	450.0	60.6	100.9	0.0
	3	Soybean	6.5	13.1	21.7	0.0
	4	Peanut	0.0	8.2	13.6	1,561.9
	5	Greenpeal	4.3	1.8	4.8	0.0
	6	Cassava	79.6	17.9	59.5	5,686.1
	7	Sweet Potato	13.6	2.6	8.7	831.7
Plantation	8	Sugar Palm	5.7	5.7	5.7	0.0
	9	Cloves	15.9	8.3	11.5	1,217.6
	10	Cashew Nut	9.4	1.6	10.7	2,889.1
	11	Cacao	57.8	24.8	49.9	0.0
	12	Rubber	291.6	72.2	266.3	0.0
	13	Cinnamon	0.2	0.1	0.1	0.0
	14	Coconut	376.8	82.3	156.3	0.0
	15	Palm Oil	381.8	300.1	472.5	0.0
	16	Coffee	148.4	29.0	112.7	0.0
	17	Pepper	22.6	9.9	9.4	0.0
	18	Nutmeg	1.0	0.2	0.5	306.0
	19	Areca Nut	10.4	7.0	10.7	0.0
	20	Sugar Cane	166.7	17.4	48.7	0.0
	21	Tea	18.6	5.4	5.1	0.0
	22	Tobacco	13.3	4.7	12.8	0.0
Horticulture	23	Shallot	7.0	3.9	13.0	1,492.8
	24	Onion	0.2	0.1	0.3	39.5
	25	Potato	12.3	4.1	8.6	1,979.7
	26	Cabbage	7.5	2.0	6.7	964.2
	27	Tomato	4.0	1.1	1.5	1,134.5
	28	Cayenne Pepper	8.5	4.8	9.6	2,441.8
	29	Chive	32.7	0.0	0.0	730.3
	30	Cauliflower	1.3	0.7	1.1	206.1
	31	Chinese Cabbage	14.2	2.5	7.7	1,068.5
	32	Carrots	2.1	0.6	0.8	440.0
	33	Radish	0.2	0.2	0.0	22.7
	34	Red Bean	0.5	0.2	0.5	0.0
	35	Long Bean	1.8	3.0	5.0	1,325.4
	36	Chili	14.0	3.6	4.7	1,804.1
	37	Paprika	0.0	0.0	0.0	2.0
	38	Terung	7.1	6.1	4.8	0.0
	39	Bean	0.9	0.7	1.6	465.3

	40	Cucumber	2.9	2.6	2.4	1,025.7
	41	Kangkung	3.7	0.0	0.0	266.8
	42	Spinach	2.2	0.9	1.9	462.1
	43	Melon (Tree)	0.0	0.1	0.2	88.9
	44	Watermelon (Tree)	0.0	1.6	326.2	412.7
	45	Cantaloupe (tree)	0.0	0.1	0.2	78.1
	46	Melinjo (Tree)	0.4	0.1	0.4	128.1
	47	Petai (Tree)	0.3	0.1	0.3	79.2
	48	Jengkol (Tree)	0.1	0.0	0.1	21.4
	49	Avocado (Tree)	1.6	0.0	0.0	21.0
	50	Starfruit (Tree)	0.1	0.1	0.2	23.9
	51	Duku/ Langsat (Tree)	0.0	0.0	0.0	350.5
	52	Durian (Tree)	0.5	0.6	0.5	252.8
	53	Guava (Tree)	0.5	0.1	0.5	146.3
	54	Pink Water (Tree)	0.1	0.0	0.0	80.3
	55	Orange (Tree)	1.9	0.8	2.0	387.0
	56	Mangga (Tree)	1.8	0.7	1.9	439.3
	57	Mangoesteen (tree)	0.0	0.0	0.0	35.7
	58	Jackfruit (Tree)	0.9	0.2	1.1	200.7
	59	Pineaple (Tree)	1.1	0.3	1.7	212.4
	60	Papaya (Tree)	1.1	0.3	0.8	468.0
	61	Banana (Tree)	16.8	2.9	19.0	5,157.9
	62	Rambutan (Tree)	0.9	0.2	1.3	312.9
	63	Salak (Tree)	2.0	0.1	0.0	745.5
	64	Sapodilla (tree)	0.1	0.0	0.0	1.6
	65	Passionfruit (tree)	0.0	0.0	0.0	22.7
	66	Soursop (Tree)	0.2	0.1	0.1	23.4
	67	Apple (Tree)	0.1	0.0	0.0	104.8
	68	Grape (Tree)	0.1	0.0	0.0	3.8
		Total	3,674	881	2,349	65,703

Section 4 Duckweed demand forecast

The forecasted fish production per province is calculated using the following approach:

- For the years 2005 until 2011 provincial data were obtained through the BPS (2014). The total fresh water fish production for each province (in kt/year) was determined using the following formula:
  ∑Fish = ∑ ([brackish pond] + [freshwater pond] + 50% * ([floating fish cage] + [fish cage]) + [rice fields]). The 50% value refers to our assumption that half of all cage fish comprises fresh water production.
- Using the linear trend formula (Excel) of the 2005-2011 data, the 2015-2035 forecasts were made on fresh water fish productions (Table A6.4).

Table A6.4 Forecasted fresh water fish production per province (kt/year)

Province	2005	2006	2007	2008	2009	2010	2011	2015	2020	2025	2030	2035
Aceh	24	32	35	43	38	46	34	51	62	73	84	94
Sumut	42	39	51	79	75	92	100	145	200	255	309	364
Sumbar	25	34	48	72	70	83	112	159	227	295	364	432
Riau	26	26	27	26	35	38	51	59	78	97	116	134
Jambi	9	10	13	16	20	26	28	41	58	75	92	109
Sumsel	73	90	102	119	169	199	264	359	512	665	817	970
Bengkulu	7	8	9	13	14	18	33	41	59	78	96	115
Lampung	138	173	187	184	102	87	107	111	124	137	150	163
Babel	1	1	1	1	1	2	2	3	4	5	6	7
Kepri	0	1	1	0	0	0	3	2	4	5	6	7
DKI	7	3	4	5	2	15	7	12	16	20	23	27
Jabar	251	287	309	351	366	529	594	777	1,058	1,339	1,620	1,901
Jateng	84	90	108	125	135	162	225	281	387	493	599	705
Yogya	9	10	12	15	18	40	44	64	95	126	156	187
Jatim	206	152	159	164	167	243	298	315	398	481	565	648
Banten	21	22	28	28	29	71	74	104	150	196	242	288
Bali	4	4	4	7	5	7	11	13	18	23	28	33
NTB	14	19	26	39	35	48	102	124	183	243	303	362
NTT	0	3	1	1	1	2	2	3	3	4	5	5
Kalbar	5	6	7	13	13	22	24	36	53	70	87	103
Kalteng	4	5	4	6	11	19	25	34	52	69	86	103
Kalsel	11	11	14	19	38	54	61	95	142	188	235	282
Kaltim	37	29	48	47	49	74	71	99	133	167	202	236
Sulut	10	7	12	14	14	21	45	51	75	99	123	147
Sulteng	12	12	21	13	17	28	47	55	78	102	126	150
Sulsel	138	108	302	277	238	542	609	870	1,266	1,661	2,057	2,452
Sultra	22	12	11	25	31	51	59	82	120	157	194	232
Gorontalo	2	2	2	3	3	6	8	11	16	20	25	30
Sulbar	-	14	5	11	15	17	19	29	42	55	68	80
Maluku	1	0	0	0	1	1	2	2	3	3	4	5
Maluku	1	0	0	0	1	2	2	3	4	6	7	9

Utara												
Papua Barat	0	6	0	1	1	1	4	6	9	13	16	19
Papua	2	2	2	2	2	2	4	4	5	7	8	9
Total	1,185	1,217	1,557	1,720	1,718	2,548	3,070	4,041	5,634	7,226	8,819	10,411

To forecast the yearly duckweed demand consumption, the provincial freshwater fish forecasts were then multiplied by 60%, corresponding with the combined tilapia and carp fraction (Dey et al. 2005; FAO 2010). These forecasts were then multiplied by a daily duckweed consumption of 450 g fresh duckweed/kg fish/d (Hassan & Edwards 1992; El-Shafai et al. 2004), multiplied by 365 days/year and divided by 1000 to convert to million t/year (Table A6.5).

Table A6.5 Forecasted duckweed demand per province (Million t/year)

Province	2015	2020	2025	2030	2035
Aceh	5	6	7	8	9
Sumut	14	20	25	30	36
Sumbar	16	22	29	36	43
Riau	6	8	10	11	13
Jambi	4	6	7	9	11
Sumsel	35	50	65	81	96
Bengkulu	4	6	8	9	11
Lampung	11	12	14	15	16
Babel	0	0	0	1	1
Kepri	0	0	0	1	1
DKI	1	2	2	2	3
Jabar	77	104	132	160	187
Jateng	28	38	49	59	70
Yogya	6	9	12	15	18
Jatim	31	39	47	56	64
Banten	10	15	19	24	28
Bali	1	2	2	3	3
NTB	12	18	24	30	36
NTT	0	0	0	0	1
Kalbar	4	5	7	9	10
Kalteng	3	5	7	8	10
Kalsel	9	14	19	23	28
Kaltim	10	13	16	20	23
Sulut	5	7	10	12	15
Sulteng	5	8	10	12	15
Sulsel	86	125	164	203	242
Sultra	8	12	15	19	23
Gorontalo	1	2	2	3	3
Sulbar	3	4	5	7	8
Maluku	0	0	0	0	1
Maluku Utara	0	0	1	1	1

Papua Barat	1	1	1	2	2
Papua	0	1	1	1	1
Total	398	555	712	869	1,026

Section 5 Numeric values of P and Compost demand per province

Table A6.6 Calculated actual and recommended (recom) P and compost demand for food crops, plantation and horticulture and indicated group of provinces for 2015, 2025 and 2035

Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
Sumatra (provinces 1-10)	P-demand (kt/y)	2015	Actual	38	287	7	332
			Recommended	64	383	15	462
		2025	Actual	45	401	10	455
			Recommended	75	535	19	630
		2035	Actual	50	514	12	576
			Recommended	84	687	24	795
	Compost demand (kt/y)	2015	Actual	6,151	126	3,670	9,947
			Recommended	9,772	252	7,341	17,365
		2025	Actual	6,613	176	4,870	11,659
			Recommended	10,506	353	9,741	20,599
		2035	Actual	7,369	226	6,071	13,666
			Recommended	11,707	453	12,141	24,301
Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
Java (province 11-16)	P-demand (kt/y)	2015	Actual	76	35	13	123
			Recommended	133	61	26	220
		2025	Actual	89	48	17	154
			Recommended	157	85	34	276
		2035	Actual	99	62	21	182
			Recommended	174	109	42	326
	Compost demand (kt/y)	2015	Actual	10,914	506	7,847	19,267
			Recommended	17,166	1,011	15,695	33,872
		2025	Actual	11,733	706	10,413	22,853
			Recommended	18,455	1,412	20,826	40,694
		2035	Actual	13,075	907	12,979	26,961
			Recommended	20,566	1,813	25,958	48,338
Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
Bali, NTT, NTB (province 17-19)	P-demand (kt/y)	2015	Actual	11	9	1	22
			Recommended	22	17	3	42
		2025	Actual	13	12	2	27
			Recommended	25	24	4	53
		2035	Actual	15	15	2	33
			Recommended	28	31	5	64
	Compost demand	2015	Actual	1,495	818	941	3,254
			Recommended	2,412	1,636	1,881	5,930
		2025	Actual	1,607	1,143	1,248	3,998

	(kt/y)		Recommended	2,594	2,285	2,497	7,375
		2035	Actual	1,791	1,467	1,556	4,814
			Recommended	2,890	2,934	3,112	8,936
Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
Kalimantan (province 20-23)	P-demand (kt/y)	2015	Actual	12	116	2	130
			Recommended	18	144	3	165
		2025	Actual	14	163	2	179
			Recommended	22	201	4	226
		2035	Actual	16	209	3	227
			Recommended	24	258	5	287
	Compost demand (kt/y)	2015	Actual	1,989	7	736	2,731
			Recommended	2,980	13	1,472	4,465
		2025	Actual	2,138	9	976	3,124
			Recommended	3,204	19	1,953	5,176
		2035	Actual	2,383	12	1,217	3,612
			Recommended	3,570	24	2,434	6,028
Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
Sulawesi (province 24-29)	P-demand (kt/y)	2015	Actual	18	36	3	57
			Recommended	33	62	5	100
		2025	Actual	22	50	3	75
			Recommended	39	86	7	132
		2035	Actual	24	64	4	93
			Recommended	43	111	9	163
	Compost demand (kt/y)	2015	Actual	2,439	940	1,352	4,730
			Recommended	3,711	1,880	2,704	8,295
		2025	Actual	2,622	1,313	1,794	5,728
			Recommended	3,990	2,626	3,588	10,203
		2035	Actual	2,922	1,686	2,236	6,843
			Recommended	4,446	3,371	4,472	12,289
Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
Papua, Maluku (province 30-33)	P-demand (kt/y)	2015	Actual	1	9	1	11
			Recommended	3	16	1	20
		2025	Actual	2	13	1	15
			Recommended	3	22	2	27
		2035	Actual	2	16	1	19
			Recommended	3	29	2	34
	Compost demand (kt/y)	2015	Actual	282	252	258	792
			Recommended	503	503	515	1,522
		2025	Actual	303	351	342	997
			Recommended	541	703	684	1,927
		2035	Actual	338	451	426	1,216
			Recommended	602	902	852	2,357
Region	Resource	Year		Food Crops	Plantation	Horticulture	Total
[illegible]	P-demand	2015	Actual	157	492	26	675

	(kt/y)		Recommended	273	683	53	1,009
		2025	Actual	184	686	35	906
			Recommended	321	953	70	1,344
		2035	Actual	206	881	43	1,130
			Recommended	357	1,224	87	1,668
	Compost demand (kt/y)	2015	Actual	23,269	2,649	14,804	40,721
			Recommended	36,544	5,297	29,607	71,448
		2025	Actual	25,017	3,699	19,644	48,359
			Recommended	39,289	7,397	39,288	85,975
		2035	Actual	27,878	4,749	24,485	57,112
			Recommended	43,783	9,498	48,969	102,250

Section 6 Discrepancy between calculated and FAOSTAT data

In paragraph 6.4.1.1 of Chapter 6 several reasons for the observed discrepancy between calculated and FAOSTAT values were introduced. In the current Section, there are further elaborated:

1. *Possible incomplete data used by FAOSTAT*: Discussions with the FAO experts in Indonesia (FAO 2014b) showed that the basis for the FAO reports is the Indonesian Fertilizer Association datasheet (IFPA 2013). Analysis of this datasheet suggests that not all private estates are included, as the contribution of private estates for use of SP36 (superphosphate, 36% P_2O_5) is less than 0.5% of the total SP36 use (IFPA 2013). Because 59% of all palm oil plantations (MoA 2014) are privately owned and palm oil is one of the major contributors to the P demand (Table 2 Chapter 6) a much higher value would be expected. When the palm oil P-demand from Table 2 (Chapter 6) is excluded a total P-demand of 310 t/year P is calculated. This value is closer to the reported FAO value of 252 t/year P;
2. *Difference in used recommended and actual fertilizer rates in calculations*. In this study the used recommended fertilizer rates for N, P and K for rice are respectively 105, 12 and 39 kg/ha/year, based on recommendations by Padjajaran University (2013). The FAO (FAO 2005) recommends an N, P and K dosage of respectively 60-100, 15-25 and 4-35 kg/ha/year, whereas (Bambang 2011) recommends values of 70, 20, 14 kg/ha/year. Actual demands were calculated (Figure 2 of Chapter 6) using the percentage of actual inorganic compared to the recommended value based on Irawan et al. (2012). Different (recommended) starting points therefore result in different calculated actual quantities. Actual input demand depends on a variety of factors, among others, soil characteristics, actual nutrient uptake by the plant, combination of fertilizers used and budget availability (Janssen & Guiking 1990; Buresh & Witt 2008; Dobermann & Fairhurst 2000). Consequently, applied correction factors of Irawan et al. (2012) are not static as was assumed. Irawan et al. (2012) only determined percentage of actual inorganic fertilizer rate compared to the recommended value for rice, soy and corn. Other values were assumed to use the average of reported values. For plantations the use of

fertilizer was made depended on type of plantation (SHE and LE). The impact of made assumptions requires further analysis and is excluded from the current study;

3. *Farmer's interpretation on fertilizer requirements*. Prices for KCl and SP-36 in Indonesia are higher than urea following the ministerial guideline (MoA 2012a; Kariyasa 2005). Consequently, farmers reportedly use KCl and SP36 as complementary fertilizers, whereas urea is regarded as the main fertilizer in farming (Kariyasa 2005). Although the effect was not quantified by Kariyasa (2005), this may result in a lower K and P use (as reported in FAOSTAT) compared to recommended values (and calculated actual values);
4. *Use of organic fertilizer as a source of N, P and K*. FAOSTAT only registers produced inorganic fertilizers, whereas organic fertilizers are not included. Organic fertilizers also contain N, P and K. Chicken manure is a popular organic fertilizer in Indonesia (Buresh et al. 2010). In addition, sheep manure production is reported as important as an output as meat production (Tanner et al. 2001). Parikesit et al. (2005) determined that nearly 70% of dairy farmers used manure for application on the land (as compost or as fresh manure) and to gain extra income by selling. Limited information is available on the actual amount applied (FAO 2005).The total N and P to be expected from animal manure is approximately 400 and 150 kt/year N and P (see Appendix Chapter 6, Section 7 for calculation). This amounts to respectively 11 and 16% of the recommended N and P values (Table 2 of Chapter 6). This may explain a difference between reported FAOSTAT values and calculated fertilizers demand, but it cannot fully explain the discrepancy;
5. *Use of phosphate rock*. Direct use of untreated phosphate rock as an alternative to commercial mineral fertilizers as SP or TSP is applied in some regions in Indonesia (Yusdar et al. 2007). Phosphate rock is not counted by FAO statistics (FAO 2014a) and application would therefore not be included. Phosphate rock is present in East Java, Central Java and West Java and especially applicable on acid grounds (Sajimin et al. 2001; Rochayati et al. 2003; Kasno et al. 2009).

Several studies suggest to make palm oil production more sustainable in terms of deforestation, land degradation or reuse of crop residues (Yoshizaki et al. 2013; Harmen Smit et al. 2013). Contrary, little attention is paid to fertilizer requirements in these studies. The used recommended fertilizer rate for palm trees in this study was verified with experts of the biggest palm oil plantation holders of Indonesia (Sinarmas 2014a; Sinarmas 2014b) who confirmed that the applied recommended rate of 0.25 kg P/tree/year in this study is in line with their current practice. This value of 0.25 kg P/tree/year is within the range of actual applied rates of 0.2-0.4 kg P/tree/year reported by Von Uexhill (1992), but about half of the 0.46 kg P/tree/year reported by Yoshizaki et al. (2013). The recommended P fertilization level for palm oil may lead to P accumulation in the soil in the long run as the export of P from the field with palm oil is low (FAO 2014a). When the P level of the soil increases, the recommended P fertilization should decrease

(Syers et al. 2008). Accumulation of P in the soil will depend on the fate of P in crop residues. Commonly, all fronds are left on the plantation field (Sinarmas 2014a). The harvested fruits contain a substantial (about half) amount of P applied (Von Uexhill 1992) and in case a mill is present at the plantation part of the P in fruits can be recycled again (Sinarmas 2014a; Yoshizaki et al. 2013).

Section 7 Estimation of N and P applied on land from animal manure

To estimate the N and P applied on the land (as referred to in Section 6.4.1.1 of Chapter 6), the following approach was applied:

1. Livestock is obtained from the agricultural yearbook on livestock by the Ministry of Agriculture (MoA 2010);
2. N and P excreta for dairy cows, goat, sheep, laying chicken and broiler chicken were obtained from Table 3 in Gerber et al. (2005) applying intensity class 3 in Gerber's article. The ratio between beef cattle mineral excreta and dairy cow was obtained from CBS (2009).
3. In this estimation it was assumed that 70% of all manure would be collected, based on Parikesit et al. (2005) and that all manure would be stored, allowing for composting to take place;
4. During composting N-loss takes place, mostly as a result of ammonification (de Guardia et al. 2010). The level of N-loss differs per feed stock and process conditions. Altuna (2013) compares several literatures of which the approximate average of 50% is applied in this estimation;
5. The amount of N and P applied on the land under these conditions is then calculated by: [cattle] x [excreta (kg/cattle/year] x [collection rate (%)] x [correction for N-loss].
   The values are presented in Table A6.7.

Table A6.7 Estimation of yearly N and P deposited on land per type of livestock

Type of livestock	Millions of animals	N-excreta kg/year/type	P-excreta (kg/year/type)	Collection rate	N-loss	N-total	P-total	Unit
Dairy cows	0.50	26.0	4.6	70%	50%	4.5	1.6	kt/year
Beef cattle	13.6	17.9	3.2	70%	50%	85.3	30.3	kt/year
goat	16.8	15.0	2.2	70%	50%	88.4	25.9	kt/year
sheep	10.9	15.0	2.2	70%	50%	57.3	16.8	kt/year
Layer chicken	116.2	0.5	0.1	70%	50%	18.7	9.3	kt/year
Broiler chicken	1,115.1	0.4	0.1	70%	50%	152.2	58.4	kt/year
Total						406.4	142.3	kt/year

Section 8 Potential recoverable fraction as part of the demand

Table A6.8 shows the impact of maximum and minimum applied sector growth rates on the potentially recoverable fraction of the demand. It combines:

- the 2035 values for compost and P-demand data (Table 6.3 of Chapter 6), the 2035 fresh duckweed production (Figure 6.6 of Chapter 6), the 2035 plastic consumption and production data (Table 6.4 of Chapter 6) and the 2035 paper production and consumption (Table 6.5 of Chapter 6);
- the maximum and minimum sector growth forecast (Table 6.6 of Chapter 6);
- the maximum recovery potential, as described in the text, namely P (170 kt/year), compost (20,130 kt/year), fresh duckweed (with 75,000 kt/year), recoverable plastic (8,036 kt/year) and recoverable paper (6,888 kt/year).

Table A6.8 Impact of maximum (exponential growth rates) and minimum (order 2 polynomial) growth rates on % potential recoverable resources in 2035 for Indonesia in relation to the Recommended (Recom) and Actual demand and Consumption (Cons) and Production (Prod).

Resource	P		Compost		Duckweed	Plastic		Paper	
	Recom	Actual	Recom	Actual		Cons	Prod	Cons	Prod
Baseline [a]	10%	15%	20%	35%	7%	68%	66%	30%	18%
Minimum [b]	5%	7%	12%	21%	1%	58%	61%	17%	11%
Maximum [c]	12%	19%	22%	39%	11%	162%	133%	77%	27%

[a] Baseline uses the growth forecast applied in this study; [b] Minimum potential recovery is obtained with highest growth forecast; [c] Maximum potential recovery is obtained with lowest growth forecast

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Chapter 7

An integrated approach to evaluate benefits and costs of wastewater and solid waste management to improve the living environment: the Citarum River in West Java, Indonesia

This paper is in preparation for submission as:
An integrated approach to evaluate benefits and costs of wastewater and solid waste management to improve the living environment: the Citarum River in West Java, Indonesia. Kerstens, S.M., Hutton, G., Firmansyah, I., Leusbrock, I. & Zeeman, G.

Abstract

Absence of wastewater and solid waste facilities impacts the quality of life of many people in developing countries. Implementation of these facilities will benefit public health, water quality, livelihoods and property value. Additional benefits may result from the potential recovery of valuable resources from wastewater and solid waste, such as compost, energy and phosphorus, plastics and paper.

Improving water quality through implementation of wastewater and solid waste interventions requires, among others, an analysis of (i) sources of pollution, (ii) mitigating measures and resource recovery potentials and their effect on water quality and health, and (iii) benefits and costs of interventions. We present an integrated approach to evaluate costs and benefits of domestic and industrial wastewater and solid waste interventions. To support a policy maker in formulating a cost- and environmentally effective approach, we quantified the impact of these interventions on (1) water quality improvement, (2) resource recovery potential, and (3) monetized benefits versus costs. The integration of technical, hydrological, agronomical and socio-economic elements to derive these three tangible outputs in a joint approach is a novelty, while attempts so far have focused mainly on these aspects individually.

The approach is demonstrated using the heavily polluted Indonesian Upper Citarum River in the Bandung region. Domestic interventions, applying simple (anaerobic filter) technologies were economically most attractive with a benefit cost ratio (BCR) of 3.2, but could not reach target water quality standards. To approach the target water quality, both advanced domestic (nutrient removal systems) and industrial wastewater treatment interventions were required, leading to a BCR of 2. Benefits from selling recovered resources represent here an additional driver for improving water quality and outweigh the additional costs for resource recovery facilities. While included benefits captured some of the major items, these may have been undervalued. Based on these findings, water quality interventions justify their costs and are socially and economically beneficial.

Keywords: wastewater, solid waste, water quality modelling, economic cost benefit analysis, resource recovery

7.1 Introduction

Nearly 40% of the population in developing countries lacks access to improved sanitation facilities (WHO & UNICEF, 2015), while an estimated 90% of all wastewater in developing countries is discharged untreated directly into rivers, lakes or the oceans (Corcoran et al., 2010). While access to improved sanitation facilities in South East Asia has reached 72%, Indonesia is lagging behind with only 61% having access (WHO & UNICEF, 2015). Less than 4% of Indonesia's septage sludge is delivered to a treatment plant and treated (WSP, 2013a). Moreover, Indonesia, like other developing countries, largely lacks proper solid waste management services and suffers from uncontrolled discharge of industrial wastewater (ADB, 2013a, 2014). The absence of proper domestic and industrial wastewater and solid waste facilities is associated with a number of impacts.

First, discharge of untreated sewage can lead to adverse health effects on individuals from contamination of drinking-water, contamination of irrigated crops or direct contact (Shuval, 2003). The World Bank's Water and Sanitation Program's (WSP) estimated that poor sanitation led to an economic loss of US$ 6 billion annually in Indonesia, equivalent to 2.3% of the national GDP in 2006 (Napitupulu & Hutton, 2008), with more than half of these costs being health related. Health conditions can be improved by wastewater and hygiene interventions (Montgomery & Elimelech, 2007; Waddington & Snilstveit, 2009; Malekpour et al., 2013). Surinkul & Koottatep (2009) showed that *E. coli* concentrations in canals could be substantially reduced (~4 log) by sewage collection and treatment. Currently, 95% of septic tanks in Indonesia are poorly functioning as they are not sealed (WSP, 2013b) and thus represent a risk to both public health and the environment (Baum et al., 2013). Wright et al. (2013) highlighted the public health risks associated with a combination of shallow ground water sources, on-site sanitation (e.g. pit latrines) and high population density.

Second, discharge of untreated wastewater will increase nutrient (nitrogen (N) and phosphorus (P)) and organic pollutants (Chemical oxygen demand (COD) and Biological Oxygen Demand (BOD)) loads in the exposed water bodies and consequently the surrounding environment. This may result in eutrophication and low oxygen levels in (coastal) waters as well as impact ecosystem functioning, leading to lower revenues from fisheries and tourism (Hart et al., 2002; Fulazzaky, 2010; Suharyanto & Matsushita, 2011; Suwarno et al., 2013). Domestic pollution depends on lifestyle and living conditions and varies with the type of residential areas (e.g. urban and rural areas) (Ujang & Henze, 2006; MoPW, 2011; Abu Ghunmi et al., 2011). A wide range of wastewater treatment systems exists and their feasibility can be linked to residential population density and urban/rural status (Kerstens et al., 2015) (see Appendix Chapter 7 Section 1). Not all pollution discharged in residential areas will reach the surface water and there is a positive correlation between imperviousness and urban density on pollution gradients in receiving water bodies (Hatt et al., 2004). In addition to domestic sources, industrial wastewater discharge may contribute significantly to water pollution (Fulazzaky, 2010). Management of industrial pollution includes pollution prevention, resource recovery and end-of-pipe wastewater treatment (Orhon et al., 2009). Applicable wastewater treatment technologies depend on type of industry, biodegradability, toxicity, robustness, and effluent standards (Orhon et al., 2009). Treatment combinations of physical pre-treatment, anaerobic and/or aerobic technologies have been successfully and widely applied for textile, food & beverage and paper and pulp wastewater (Thompson et al., 2001). To allow effluent reuse, post-treatment including filtration, chemical oxidation, reverse osmosis and disinfection is needed (Gozálvez-Zafrilla et al., 2008). A challenge in managing industrial pollution prevention in Indonesia is limited availability and contradicting data on the status of industrial pollution (De Vries, 2012). Water quality is further affected by agricultural activities, as a result of fertilizer use, aquaculture and livestock emissions (Abery et al., 2005; Suwarno et al., 2013). Improvement of the environment also requires solid waste management (Ersoy et al., 2008), distinguishing minimum interventions (all waste is landfilled) and interventions in which recoverable waste (e.g. organic waste, paper and plastic) is *reduced, reused and recycled* (3R) (Antonopoulos et al., 2014). Here, the selection of solid waste interventions can be linked to residential features (Kerstens et al., 2015).

Third, the value of recoverable resources from wastewater and solid waste, such as energy, water, organics, nutrients, plastic and paper is frequently neglected, whereas the sale of recovered resources can assure long-term operational and financial sustainability (Aye & Widjaya, 2006; Kerstens et al., 2009; Murray & Ray, 2010; Aprilia et al., 2012). The potential demand for recovered resources depends on agricultural activities (e.g. compost for horticulture, phosphorus as fertilizer, duckweed for aquaculture) and possibilities to replace conventional production processes by processes using recyclables (paper and plastics) (Kerstens, Priyanka, et al., *in preparation*).

Finally, the absence of sanitation, wastewater and solid waste facilities may accrue socio-economic impacts, such as travel and waiting time for community or public toilet facilities, loss of social capital and equity and decreased property values (Alam, 2008; Fulazzaky, 2010; Winara et al., 2011; Hutton, 2013).

Thus, implementation of wastewater and solid waste interventions benefits public health, the (aquatic) environment, resource conservation, the economy and people's welfare. However, given that implementation of interventions involves costs in the form of investments, operation and maintenance of the facilities, policy makers need to understand the outcomes (benefits) of major actions in relation to their costs before making choices (Ward, 2012). For example, wastewater interventions to meet a certain water quality may differ from those that aim (a) to provide basic access (Baum et al., 2013) or (b) to recover valuable resources from wastewater. The Benefit Cost Ratio (BCR) describes benefits of intervention (e.g. health, social, resource recovery) relative to its costs (implementation and operation). Given that benefits may require a long time to manifest and planned infrastructure are designed for long lifetimes, benefit-cost analysis should use a time horizon of at least 20 years (Ng et al., 2014). A demonstrated BCR of one or more – indicating a return on investment of at least 1.0 given the discount rate used - can feed into advocacy efforts to raise funding from governments and households, and can even convince the private sector to invest if there are financial returns (Hutton, 2013).

Individual cause-effect relationships have already been established in order to evaluate the costs and benefits of different interventions to improve water quality and improve solid waste management, such as: (1) the effect of discharging a pollution load on the quality of receiving water (e.g. Hatt et al., 2004; Suharyanto & Matsushita, 2011)), (2) the effect of sanitation intervention on improvement of public health (e.g. Malekpour et al. (2013)), (3) the effect of implementation of wastewater interventions on discharged pollution loads (e.g. Suwarno et al. (2013)), (4) economic losses as a result of poor sanitation (e.g. Hutton (2013)), and (5) technical and financial feasibility of wastewater and solid waste technologies (e.g. Kerstens et al., (2015)). However, no integrated framework exists in the scientific literature that quantifies the effect of applicable wastewater and solid waste interventions on (1) water quality, (2) resource recovery potential, and (3) monetized benefits and costs. This paper therefore proposes to use a combination of methods that describe these individual cause-effect relationships, and synthesize them to produce these three tangible outputs. This multi-methods approach then allows policy makers to make well-informed choices in wastewater and solid waste planning.

The developed approach can be used on any river basin or delta. In this paper, the Upper Citarum River is used as a case study because of its very low water quality combined with its impact on the life of millions of people downstream of where major pollution is taking place. The Upper Citarum basin (Figure 7.1) is located in the center of the West Java, Indonesia (see also Appendix Chapter 7, Section 2, Figure A7.1 and A7.2) and forms the upstream area of Saguling reservoir. The Citarum River was listed among the 10 most polluted places in the world as a result of domestic and industrial pollution (Gannon, 2013). The area is known for its textile production that accounts for over 80% of the industrial water consumption (Kerstens et al., 2013). The greater Jakarta area houses approximately 25 million people, and currently receives 40% of its domestic, municipal and industrial water from the Citarum River, which is projected to increase to 75% in the coming decades (MoPW, 2011). In addition, the Citarum River and its reservoirs are used for agriculture, aquaculture and leisure (Fulazzaky, 2010).

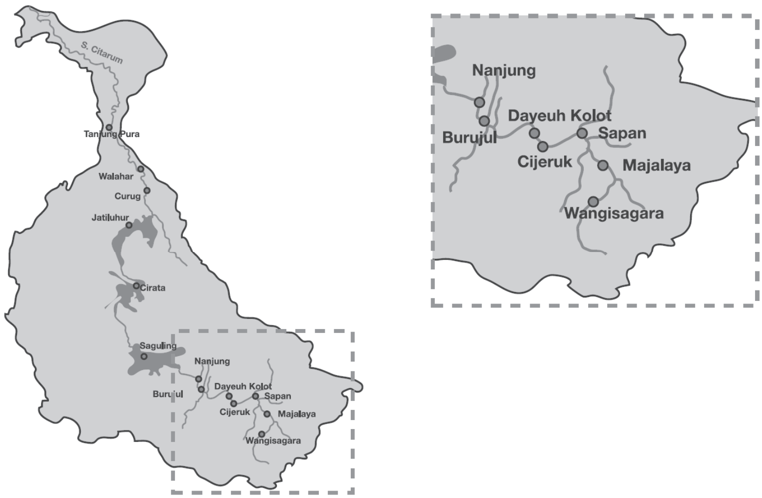

Figure 7.1 Location of the Upper Citarum River basin (in box) within the Citarum basin

7.2 Materials and Methods

To assess the impact of wastewater and solid waste interventions on water quality and estimate resource recovery and economic returns, the following six consecutive steps were formulated (Figure 7.2). In step 1 the river water quality at different locations was collected. This information was used as a baseline to determine the impact of different types of interventions. In step 2 the sources of pollution COD, BOD, N and P per sector (domestic, industrial and agricultural) were determined. An additional assessment on the relative contribution per sector was performed considering variations in the pollution load reaching the surface water with different urban areas (Hatt et al., 2004) and the status of industrial pollution control (De Vries, 2012). In step 3 wastewater and solid waste interventions were defined and their associated costs estimated, based on Kerstens et al. (2015). The impact on pollution loads discharged to the environment and the associated costs were further analyzed by varying treatment technologies and the percent of households switching from a septic tank to a sewer system connection. In step 4 the impact of different interventions on water quality was determined using a river basin simulation software (RIBASIM) (Deltares, 2009). In step 5, five different benefits were monetized: health, access time, water quality, environment and revenues from resource recovery (Winara et al., 2011; Kerstens et al., 2015). In step 6 the benefits and costs were compared over a 20 year period to estimate the benefit-cost ratios. In this final step also a sensitivity analysis was performed to determine the impact of reduced health, welfare and revenues from recovered resources and of different capital lifespan on the BCR. A description of the individual steps and method for data collection is further illustrated using the Upper Citarum River as an example.

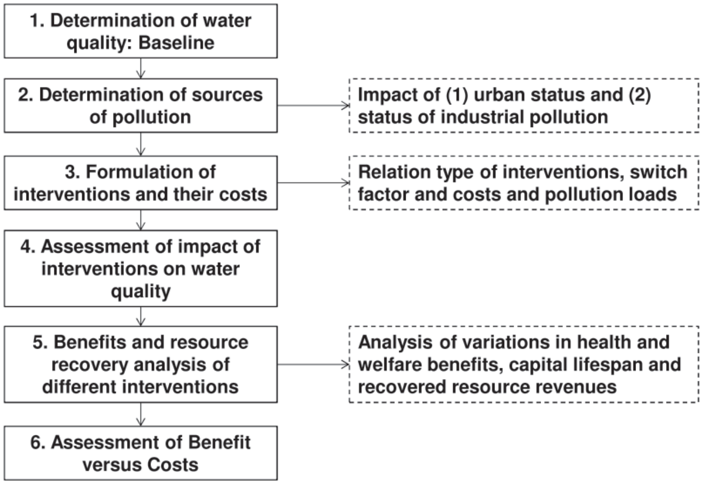

Figure 7.2 Approach applied to determine the BCR of interventions. (Dashed) blocks show activities for which a sensitivity analysis was performed

7.2.1 Step 1: Determination of water quality in Upper Citarum River

Water quality data for COD, BOD, N and P for the period 2001-2009 in the upper Citarum River at Wangisagara, Sapan, Cijeruk, Deyuekholot and Nanjung (Figure 7.1) was obtained through the West Java Regional Environmental Agency (BPLHD, 2010a; BPLHD West Java, 2011).

7.2.2 Step 2: Determination of sources of pollution

Three sources of pollutions were distinguished and assessed for 2010 and 2030, being (A) Domestic, (B) Industrial and (C) Agricultural (Table 7.1).

A. Domestic pollution:

Domestic pollution was determined in five steps.

1. Determination of specific per person pollution loads: Domestic specific water consumption rates followed the Indonesia guidelines (MoPW, 2011) for 6 categories of urban area: (1) metropolitan (> 1 million people), (2) large town (500,000-1 million people,) (3) medium town (100,000-500,000 people), (4) small town (20,000-100,000 people), (5) village (3,000 – 20,000 people) and (6) rural (<3,000 people). An 80% return factor was used to estimate wastewater production from consumed water (DKI, 2005). Metropolitan specific pollution loads were based on Kerstens et al. (2015).
2. Correction of pollution load with varying types of urban status: The relation between urban category and pollution loads was reflected using the study of Abu Ghunmi et al. (2011) applying a greywater pollution load decrease between urban metropolitan and rural areas of

30% of COD and N and 50% of P, while for urban categories in between metropolitan and rural areas these were made relative to water consumption data (Table 7.1). Because of lack of detailed data, urban and rural black water pollution load rates were assumed the same.

3. Correction of pollution reaching surface water bodies: Baseline pollution correction coefficients (included in Table 7.1) were based on Hatt et al. (2004) and were 100% (metropolitan and large towns), 92% (medium town), 83% (small town), 74% (village) and 65% (rural areas). Thus, only 83% of pollution generated in a small town is expected to reach the surface water. As specific information on these coefficients was lacking for the Upper Citarum Basin, two alternative scenarios were compared, being (1) where 100% of pollution entered the surface water, and (2) where half of the baseline value entered the surface water (i.e. 50% for metropolitan and large towns, 46% for medium town, 42% for small towns, 37% for village and 33% for rural areas).
4. Determination of pollution loads reaching the surface water for 2010 and 2030: Total specific pollutions loads per location reaching the surface water were calculated applying the specific pollution loads (combining step 2 and 3 above) on population developments obtained from the Java Spatial Model (JSM). JSM shows the population development for each urban category between 2010 and 2030 (MoPW, 2011).
5. Determination of the number of people with access to wastewater facilities in 2010: The pollution loads reaching the water bodies were corrected for interventions already in place. The 2010 wastewater access data were obtained from the statistical bureau of Indonesia (BPS) and were determined as 52%. 490,000 people were connected to the *Bojong Soang* WWTP (pond systems) in Bandung (Bojong Soang, 2012).

B. Industrial pollution:

838 industries in the catchment area were categorized by location and type (Table 7.1) and water consumption (m^3/d) in which data on ground and surface water consumption were obtained from the West Java provincial agency for Energy and Mineral Resources (ESDM, 2009) and provincial agency for Water Resources Management (PSDA, 2010). Pollution loads were determined by effluent flow (using 80% return factor) and effluent concentrations (Table 7.1). Because reliable industrial pollution data is lacking (De Vries, 2012), an impact analysis was performed (Table 7.2). A distinction is made between 1) a best case scenario, 2) a baseline scenario and 3) a worst case scenario. These scenarios vary in terms of removal efficiency and percentage of industry having a WWTP (Table 7.2). COD removal efficiencies in the best case followed self-reported COD removal efficiencies by industries, whereas the worst case effluent COD values followed externally measured COD removal efficiencies (De Vries, 2012). N and P are not measured by industries and presented values were assumed, based on Orhon et al. (2009). BPLHD (2010) reports that 80% of the textile industries comply with the effluent standards, whereas the environmental office in nearby Cimahi mentions 3% (KNLH, 2010). Therefore, the baseline case

assumes that 80% of the largest industries (consumption > 2,000 m^3/d) treat their wastewater, while with decreasing water consumption this percentage decreases with a minimum of 25% (Table 7.2).

C. Agricultural pollution:

The 2010 and 2030 water demand for irrigation was based on MoPW (2011). Pollution discharged (Table 7.1) for rice and non-rice crops were based on BWRP (2000).

Table 7.1 Basis for applied Domestic (A), Industrial (B) and Agricultural (C) pollution reaching the surface water

A. Domestic per capita pollution loads reaching surface water [a]							*B. Industrial concentrations in effluent per type of industry*				
Urban Category	*Water use*	*COD*	*BOD*	*TN*	*TP*	*coliform*	Type of Industry	*COD*	*BOD*	*TN*	*TP*
	l/cap/d	g/p/d				1/100 ml		mg/l			
1. Metropolitan	190	82.2	41.1	12.3	2.1	1 x10^8	Food & beverage [b]	5,000	3,000	80	30
2. Large town	170	81.0	40.5	12.3	2.0	1 x10^8	Paper [c]	4,000	1,500	20	10
3. Medium town	150	73.5	36.7	11.3	1.9	1 x10^8	Pharma-ceutical [d]	5,000	1,500	127	25
4. Small town	130	65.3	32.7	10.2	1.7	1 x10^8	Rubber [d]	7,340	4,400	1,100	220
5. Village	100	56.9	28.5	9.1	1.5	1 x10^8	Textile [e]	1,350	450	60	20
6. Rural	30	47.3	23.7	7.9	1.3	1 x10^8	Others [d]	280	168	42	8

C. Agricultural pollution loads (g/Yield.ha) [f]					
Type of crops	*COD*	*BOD*	*TN*	*TP*	*Coliforms*
Rice	45	22.5	21.5	6.5	0
Non-rice food crops	34	17	4.6	0	0

[a] Based on Kerstens et al. (2015), Abu Ghunmi et al. (2011), and Hatt et al. (2004); [b] Based on data obtained by authors from Food & Beverage (dairy, brewery) in Indonesia and [c] Values depend on type of paper and pulping process and range from typically 1,500 to over 20,000 mg/l COD (Arantes & Milagres, 2007; Orhon et al., 2009). Applied values are based on experience of authors for Pulp and Paper South East Asia; [d] based on BWRP (2000), [e] Textile industry data were determined based on actual measurements of 21 textile industries in project area (De Vries, 2012) and verified with Orhon et al. (2009); [f] based on BWRP (2000)

Table 7.2 Defined scenarios to determine the impact of industrial pollution loads by varying (1) removal efficiencies and (2) availability of WWTP based on size of water intake [a].

Scenario	1. % Removal efficiency				2. % industries with WWTP per size of water intake (m^3/d)				
	COD	BOD	TN	TP	0-100	100-500	500-1,000	1,000-2,000	>2,000
Best case	90	95	90	50	35	35	60	80	90
Baseline	65	69	65	36	25	25	50	70	80
Worst case	40	42	40	22	15	15	40	60	70

[a] Table 7.2 shows for example that in the best case scenario, 90% of the industries with a water consumption exceeding 2,000 m^3/d have a WWTP and removal efficiencies are 90% (COD), 95% (BOD), 90% (TN) and (50% (TP)

7.2.3 Step 3: Formulation of interventions and their costs

Domestic interventions:

Selection of type of domestic WWT facilities (Table 7.3) was based on the residential features following Kerstens et al. (2015). For off-site systems three scenarios were compared to identify the effect on the surface water quality and cost:

1. Simple Technology (ST): Anaerobic filter is applied for medium centralized systems and a conventional activated sludge (CAS) for centralized systems;
2. Advanced Technology (AT): Medium central and central systems apply a CAS with additional N, P removal;
3. Resource Recovery technology (RR): Comprising Aerobic Granular Sludge (AGS) system with sludge digestion, P-recovery as struvite and composting of produced sludge. The removal efficiencies of AT and RR are the same.

Associated investment and operational costs were based on Kerstens et al. (2015) (see Table A7.1 of the Appendix Chapter 7, Section 1). The effects on discharged pollution loads reaching the surface water and associated investment costs of a 25%, 50% and 75% switch of households currently applying on-sites system to an off-site system were compared.

Table 7.3 WWT system selection based on (1) population density and (2) urban/rural category. Removal efficiencies of Simple technologies (ST), Advanced Technologies (AT) and Resource Recovery (RR) technologies for COD, BOD, TN, TP and coliforms are based on Kerstens et al. (2015)

<table>
<tr><th rowspan="3">System</th><th colspan="2">Criteria for use[a]</th><th colspan="10">Applied removal efficiencies per type of technology</th></tr>
<tr><th rowspan="2">Residential population density (pp/ha)</th><th rowspan="2">Status 2020[b]</th><th colspan="2">COD (%)</th><th colspan="2">BOD (%)</th><th colspan="2">TN (%)</th><th colspan="2">TP (%)</th><th colspan="2">Coliforms (%)</th></tr>
<tr><th>ST</th><th>AT/ RR</th><th>ST</th><th>AT/ RR</th><th>ST</th><th>AT/ RR</th><th>ST</th><th>AT/ RR</th><th>ST</th><th>AT/ RR</th></tr>
<tr><td>On-site</td><td><100</td><td>Rural/ Urban</td><td colspan="2">40[a]</td><td colspan="2">45</td><td colspan="2">15</td><td colspan="2">5</td><td colspan="2">90</td></tr>
<tr><td>CBS</td><td>>100</td><td>Rural</td><td colspan="2">80[a]</td><td colspan="2">85</td><td colspan="2">15</td><td colspan="2">5</td><td colspan="2">99</td></tr>
<tr><td>Medium Central</td><td>100-250</td><td>Urban</td><td>80</td><td rowspan="2">88</td><td>85</td><td rowspan="2">97</td><td>15</td><td rowspan="2">90</td><td>5</td><td rowspan="2">67</td><td>99</td><td rowspan="2">99.9</td></tr>
<tr><td>Central</td><td>>250</td><td>Urban</td><td>88</td><td>97</td><td>73</td><td>29</td><td>99.9</td></tr>
</table>

[a] Current users in urban areas with a residential density between 25-100 pp/ha apply on-site systems, whereas all new development will be served by medium centralized system (Kerstens et al. *in preparation).* [b] Selection criteria are formulated based on the expected population status in 2020 (mid-term)

Industrial interventions:

Three industrial wastewater treatment types were formulated based on currently applied technologies (MPS & Nijhuis, 2012) (see Appendix Chapter 7, Section 3): (1) textile wastewater using reactive dyes (typically used for traditional *batik*), apply a CAS and activated carbon for color removal, (2) textile wastewater using non-reactive dyes apply CAS followed by Dissolved Air Flotation (DAF), and (3) other industries apply pre-treatment (DAF) and CAS. Future effluent values should meet at least current standards (KNLH, 2010) defined as 80 mg/l COD, 20 mg/l BOD, 10 mg/l N and 10 mg/l P. Investment and operational costs were determined for different sizes of treatment capacities, based on available engineering cost standards (see Appendix Chapter 7, Section 3).

Municipal Solid Waste (MSW) interventions:

Solid waste system selection interventions (Table 7.4) and their costs are based on Kerstens et al. (2015) and distinguish home composting, landfilling and centralized and decentralized 3R application (see Appendix Chapter 7, Section 4).

Table 7.4 MSW system selection for Indonesia as a function of density, urban/rural status (Kerstens et al., 2016)

<table>
<tr><th rowspan="2">Type of area & density
Activity</th><th colspan="2">Rural</th><th colspan="2">Urban</th></tr>
<tr><th><25 pp/ha</th><th>>25 pp/ha</th><th><100 pp/ha</th><th>>100 pp/ha</th></tr>
<tr><td>Collection</td><td>no</td><td>yes</td><td colspan="2">yes</td></tr>
<tr><td>Disposal</td><td>no</td><td>yes</td><td colspan="2">yes</td></tr>
<tr><td>Level of 3R</td><td>Home composting</td><td colspan="2">decentralized composting and plastic/paper recovery</td><td>central digestion and composting and plastic & paper recovery</td></tr>
</table>

7.2.4 Step 4: Assessment of impact of interventions on pollution loads and water quality

A generic model package (RIBASIM) for simulating the behavior of river basins under various hydrological conditions was used to simulate the effect of different interventions on water quality development in the Upper Citarum River (Deltares, 2009; Gonenc et al., 2014). Based on pollution loads produced in each defined catchment area and resulting water flows concentrations are calculated. The RIBASIM model and defined catchment areas are further explained in Appendix Chapter 7, Section 5. The pollution loads entering the Upper Citarum River were varied, using 6 scenarios (Table 7.5).

Table 7.5 Defined intervention scenario (S1-S6); ST = Simple Technology ; AT = Advanced Technology and percentage of population served by a municipal solid waste (MSW) system

Name		Description
S1: Baseline		2010: Baseline situation
S2: No intervention		2030: Baseline case; same WWT access percentage as 2010 applied. Only correction for population growth for WWT and MSW
S3:	25% ST	2030: 100% Domestic access, use ST and 25% switch + 100% MSW
	25% AT	2030: 100% Domestic access, use AT and 25% switch + 100% MSW
	50% ST	2030: 100% Domestic access, use ST and 50% switch + 100% MSW
	50% AT	2030: 100% Domestic access, use AT and 50% switch + 100% MSW
	75% ST	2030: 100% Domestic access, use ST and 75% switch + 100% MSW
	75% AT	2030: 100% Domestic access, use AT and 75% switch + 100% MSW
S4: Industrial only		2030: Industrial WWT intervention; 100% of big (> 1, 000 m^3/d), 90% of medium (500-1,000 m^3/d), 80% small (100-500 m^3/d), and 75% of very small (< 100 m^3/d) sized industries apply intervention. Domestic WWT, MSW interventions follow S2
S5: 25-75% ST/AT		2030: Combination of scenario 3 and 4
S6[a]: 25-75% RR		2030: Same as S5, using recovery technologies for domestic, industrial effluent recycling and MSW

[a] Except for S6, where a MSW resource recovery based system is applied, all other cases apply a conventional MSW system (no resource recovery)

The output of the 2010 RISBASIM average pollutant concentrations was calibrated based on the average measured concentration (step 1).

7.2.5 Step 5: Benefits analysis of different interventions

Five economic benefits of wastewater and solid waste management improvements were defined following Hutton (2013) and Kerstens et al. (2013):

A. Health:

Averted costs of fecal-oral disease from improved on-site sanitation and wastewater management: An average disease reduction of 36% by on-site sanitation and an additional 20% by adding improved off-site facilities was applied (Moraes et al., 2003; Waddington et al., 2009; Barreto et al., 2010). The average annual health cost per 5 member family as a result of unimproved sanitation was US$ 316 (Winara et al., 2011).

Associated averted health impacts (infectious diseases and skin complaints) of less exposure during flooding events: Reported health cases during a period of several flooding events (January - March 2009) were compared to the same period in a non-flood year (January to March 2010) and was scaled to reflect all the flooded communities in the Citarum River basin, resulting in an estimated 15,000 averted cases of diarrhea in an average year (Kerstens et al., 2013). The economic value was estimated by multiplying the average number of additional cases per year by the unit cost of inpatient (hospitalized) and outpatient services, including productivity losses (Winara et al., 2011).

B. Access time:

Value of time savings from reduced travel time and/or queuing for meeting sanitation needs. An average daily gain of 115 minutes per household with an annual value of US$ 95 per household is used (Winara et al., 2011). Only the time of adults and school-aged children were included, valued at 30% and 15% of the hourly rate implied by the GDP per capita, respectively (Gwilliam, 1997). This figure was applied to the access gain afforded by on-site sanitation facilities of 45% of households for the period from 2010 until 2030.

C. Water:

Reduced drinking water treatment costs to households and industries. The total cost of water treatment (including both capital and operating costs) using surface water of a better quality source will decrease from 0.13 to 0.06 US$/m^3 (MoPW, 2011). This saving was multiplied by the assessed annual production of water from surface water sources (207 million m^3 for domestic and 70 million m^3 for industrial consumers) in 2030.

Improved fish yields from farming in downstream lakes due to improved water quality. Data collected through interviews with the regional Fisheries Office showed a decrease in fish catch of 5,000 ton/year in recent years (WSP, 2012). Fish kills in Saguling (Figure 7.1) related to discharge of untreated wastewater have been described by Hart et al. (2002) and Abery et al. (2005). By 2030 the fish capture is estimated to increase by 8,000 metric tons per year (WSP, 2012). The increase of improved water quality was assumed to account for one-third of this expected annual gain of farmed fish in the Citarum basin (Kerstens et al., 2013). A market prices of fish of 1.5 US$/kg was used (WSP, 2012).

D. Environment:

Reduced frequency of river and reservoir dredging due to improved sludge and waste management. An estimated 35 l/person/year of septic waste (WSP, 2013b) and 11% and 17% of domestic urban and rural solid waste (Ministry of Health, 2010) accumulating to nearly 500 kt/year (t=tonne) are currently discharged to the surface water and will be prevented from being disposed in the surface water in 2030 with the described interventions (Table 7.5). With a cost of dredging estimated at US$ 3.76 (MoPW, 2011) per ton of sediment (assuming no degradation), the total annual cost averted was estimated.

Rise in land prices due to improved aesthetics of riverside and lakeside real estate. Currently the Citarum riverside area is not developed due to water pollution. However, the area is expected to become a place where riverside property could be developed for inhabitants, small businesses, and tourist facilities in a situation where water quality is improved. The current agricultural land price (10.7 US$/m^2) in the vicinity of Bandung was used as a benchmark for current riverside land prices. The current market suggests that land prices can climb to 71.3 US$/m^2 in highly desirable locations (MoPW, 2011). In this study 50% of this increase is attributed to improved water quality. This value was multiplied by an estimated 50 ha of land that could be developed each year after the water quality improvements have occurred (Kerstens et al., 2013).

Averted maintenance costs of hydro-electric facilities. Improved solid waste management would avert the current costs of US$ 0.1 million (MoPW, 2011) to evacuate the solid and unmanaged sludge waste to avoid equipment damage in the hydroelectric facility (Kerstens et al., 2013) .

E. Recovery of resources:

In scenario 6, resource recovery was considered (see also Table A7.4 in the Appendix Chapter 7, Section 4):

- Off-site wastewater systems: Production of energy (sludge digestion), struvite (from centrate) and compost (digested sludge composting);
- MSW: Energy and compost production from organic waste and recovery of plastics and paper;
- Industrial wastewater: industries with a water consumption exceeding 2000 m^3/d reused 80% of the effluent, whereas for industries using 1000-2000 m^3/d this was 50%.

To compare the production (recovery) of resources with the potential demand in the Upper Citarum River catchment area in 2030, the compost, struvite, plastic and paper demand in the whole of West Java obtained from Kerstens, Priyanka, et al. (*in preparation*) was corrected for people living in the Upper Citarum River basin area. The amount of recycled water from industries was compared to the total domestic and industrial water demand in 2030 in the catchment area (MoPW, 2011). Energy production from digestion is compared to the energy demand for domestic wastewater treatment in the area applying aerobic granular sludge technology (Kerstens et al., 2015).

7.2.6 Step 6: Assessment of Benefits versus Costs

To relate benefits and costs to either wastewater or solid waste interventions, BCR's were presented separately. To analyze the individual impact of domestic, industrial and resource recovery interventions the BCR of scenarios S3: (50% ST and S3: 50% AT), S4: (Industrial interventions only), S5: (50% ST; S5: 50% AT) and S6 (50% RR) were determined (see also Table 7.5). A sensitivity analysis was performed in which input values that have the highest anticipated impact were varied: (1) health and access time benefits reduced from 100% to 50%, (2) lifespan of all wastewater and solid waste facilities varied from 20 years to 15 and 40 years, and (3) resource selling price reduced to half baseline values (Hutton, 2013; Kerstens et al., 2013). Health and access time benefits were all attributed to domestic intervention. Water quality and environmental benefits were attributed to the fraction of COD load discharged by domestic and industrial sources respectively.

7.3 Results

7.3.1 Water quality in Upper Citarum River

Figure 7.3 shows the average 2000-2009 water quality from upstream to downstream locations. Maximum allowable concentrations are defined in class II standard (GoI, 2001) and are COD 25 mg/l, BOD 3 mg/l and P 0.2 mg/l. From Sapan on (Figure 7.1) all measured values exceed these standards. Concentrations in several Citarum branches passing high density urban areas, show COD values approaching 500 mg/l and pathogen levels as high as 10^7 Units/100 ml (BPLHD West Java, 2011).

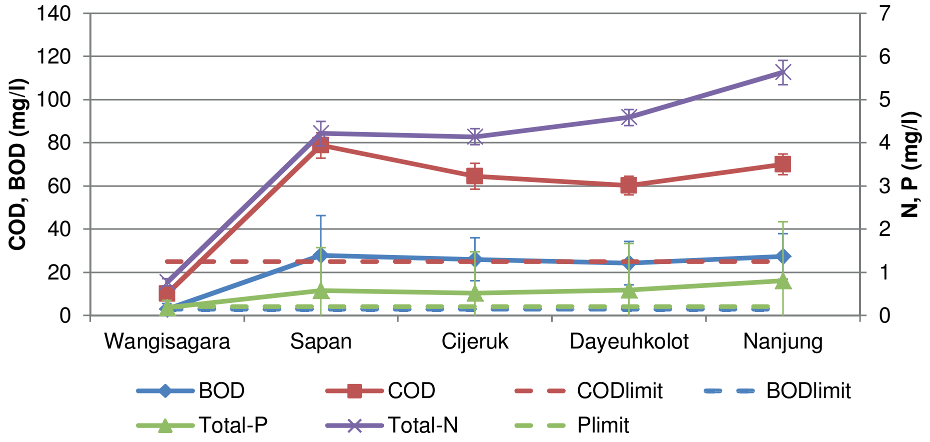

Figure 7.3 Average and standard variations of COD, BOD (primary Y-axis) and N, P (secondary Y-axis) concentrations at indicated locations in the upper Citarum (2000-2009) (BPLHD, 2010a) and COD, BOD and N limits

7.3.2 Sources of pollution

Current cumulative pollution loads in the Upper Citarum River basin of COD (585 t/d), BOD (264 t/d), TN (91 t/d) and TP (20 t/d) were determined as the baseline values (Table 7.6). The sensitivity analysis with variations in domestic pollution coefficient (Hatt et al., 2004) and performance of industries shows considerable differences with the baseline scenario (Table 7.6) with COD loads varying between 325 and 688 t/d (see also Appendix Chapter 7, Section 7).

Table 7.6 COD, BOD, TN and TP pollution loads reaching the surface water by source for the baseline scenario and varying pollution correction factors and industrial practices

Source	Scenario	COD (t/d)	BOD (t/d)	TN (t/d)	TP (t/d)
Domestic	Baseline loads	388	188	68	12
	100% reaches surface water	*440*	*213*	*78*	*14*
	Half of baseline loads reach surface water	*194*	*94*	*34*	*6*
Industrial	Baseline	163	60	6	2.6
	Best case	*98*	*33*	*4*	*2.2*
	Worst case	*215*	*80*	*8*	*3.0*
Agriculture		34	17	16	5
Total	Baseline [a] (S1)	585	264	91	20
	Minimum [b]	*325*	*144*	*54*	*13*
	Maximum [c]	*688*	*310*	*103*	*22*

[a] Total baseline values comprise domestic and industrial baseline loads + agricultural loads

[b] Total minimum values add domestic low pollution correction coefficient and Industrial best case + agricultural loads

[c] Total maximum values add domestic high pollution correction coefficient and Industrial worst case + agricultural loads

7.3.3 Effect of selected interventions on costs and pollution loads

The domestic pollution loads entering the Upper Citarum River depend on (1) the type of technology applied (simple versus advanced) and (2) the rate of current households applying on-site systems in urban areas that will switch to an off-site system (Figure 7.4). The use of advanced compared to simple technologies has a minor impact on COD removal in the range of 3-4%, but a major impact on N-removal in which a rate of 25% households switching to off-site systems leading to a 29% difference and a rate of 75% households switching to off-site systems leads to a 37% difference (Figure 7.4).

When increasing the switch factor from 25% to 75%, the additional removed COD and N increased with 5% and 1% for simple technologies and 6% and 9% for advanced technologies. BOD removal follows the COD trend, whereas P removal follows the N trend. Thus, the application of advanced technologies or a higher rate of people switching from on-site system to off-site systems mainly affects the additional nutrient removal, while organic removal is less affected. The numeric values of this analysis and further elaboration on costs of interventions and their impact on water quality are described in the Appendix Chapter 7, Section 7.

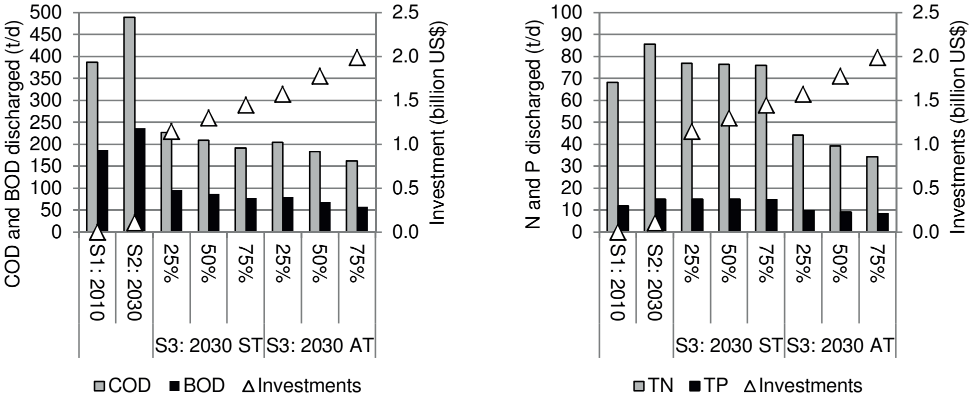

Figure 7.4 Calculated domestic COD, BOD (**left**) and N, P (**right**) pollution loads per type of intervention and their investment costs (secondary y-axis). S1 (baseline), S2 (no intervention) and S3 (domestic interventions) applying simple (ST) or advanced technologies (AT) with increasing (25%, 50% and 75%) values for urban on-site users that switch to off-site systems

The industrial pollution load amounts to 28% of the total load (Table 7.6), but industrial interventions can result in a relatively large COD reduction (35%) compared to the combined domestic and industrial COD reduction (see also Appendix Chapter 7, Section 7).

7.3.4 Effect of interventions on water quality

Figure 7.5 (A-F) shows the effect of interventions on the year round average water quality at different locations. The location names are approximate locations, as RIBASIM calculates concentrations in defined segments of a river (see Appendix Chapter 7, Figure A7.8). Without additional interventions all concentrations will increase compared to the 2010 values (Figure 7.3) with values as high as 100 mg/l of COD (Figure 7.5A). The modeled pollutant concentrations in water entering Saguling reservoir (approximate location Nanjung) are 80 mg/l COD, and 7 mg/l TN and 1 mg/l TP. When applying S3 with 50% AT (Figure 7.5B) a considerable drop in all pollution concentrations is achieved, whereas the introduction of industrial interventions result in approximately 20% COD & BOD removal and about 4% N & P removal (Figure 7.5C). The combination of these interventions (S5: 50% AT; Figure 7.5D) results in concentrations of 30 mg/l for COD, 10 mg/l for BOD, 3.4 mg/l for TN and 0.7 mg/l for TP . The maximum removal scenario (Figure 7.5F) results in values approximating the class II standard (COD < 25, BOD< 3, P<0.2 mg/l). Comparing Figure 7.5E (ST) with Figure 7.5F (AT) shows limited impact on COD or

BOD removal, while considerable extra N, P removal is shown when using advanced instead of simple domestic technologies.

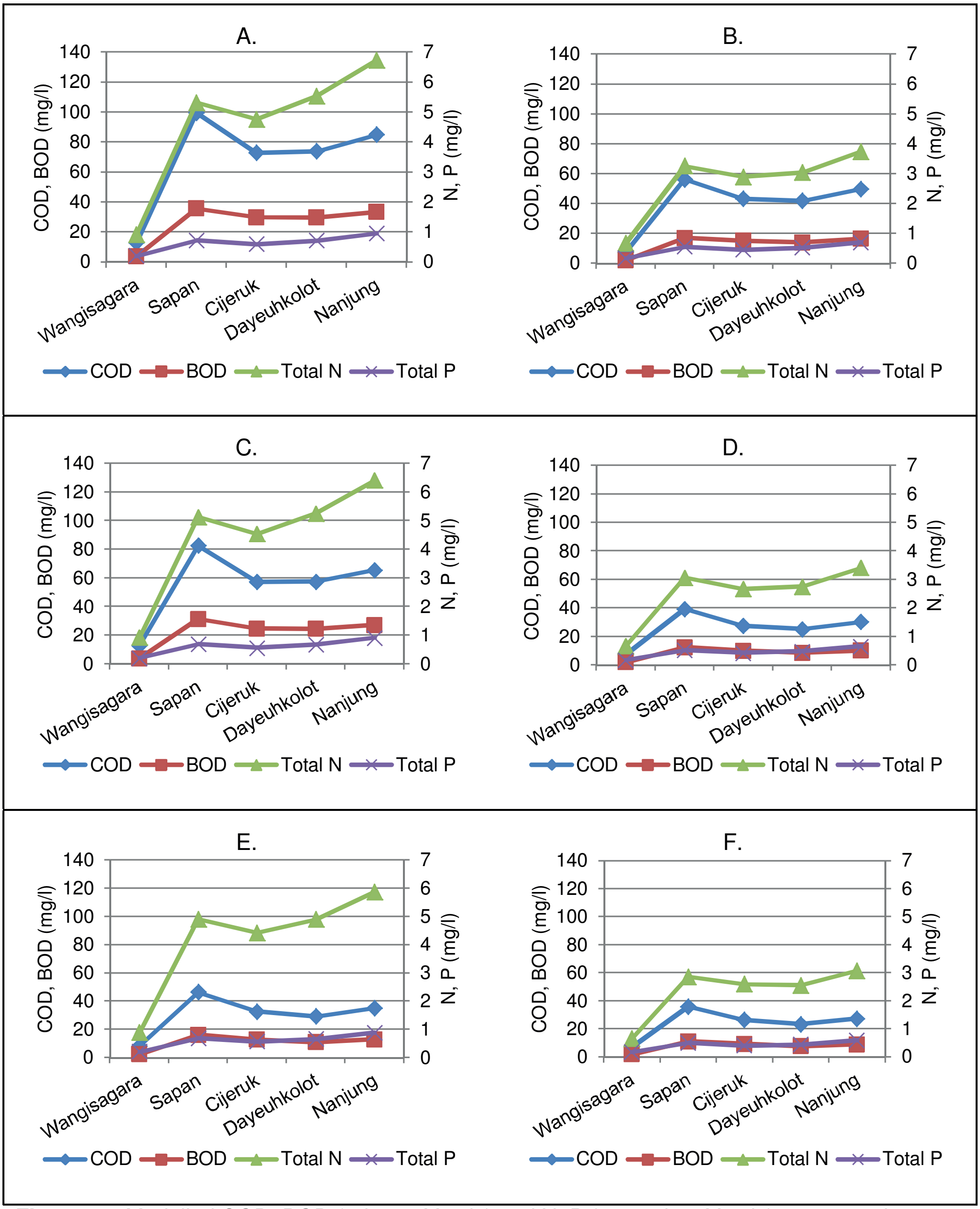

Figure 7.5 Modelled COD, BOD (primary Y-axis) and N, P (secondary Y-axis) concentrations at indicated locations in 2030 with varying switch factors % and simple (ST) or advanced technologies AT). **A**: S2, no intervention; **B**. S3: 50% AT); **C**. S4: Industrial only; **D**. S5: 50% AT; **E**. S6: 25% ST; **F**. S6: 75% AT. Limits for COD, BOD, and P are 100 mg/l, 3 mg/l and 0.2 mg/l

To reach the desired water quality levels (class II) both industrial and domestic municipal interventions are needed. In addition, the applied off-site technologies should also include N and P removal, requiring more advanced and more costly technologies (see Figure 7.4) compared to the application of only anaerobic filters.

7.3.5 Benefits of interventions

The maximum quantified economic benefits are US$ 430 million per year in which health benefits account for 39% (Figure 7.6).Health benefits largely result from reductions in fecal-oral diseases, since (1) the people without access to wastewater (on-site and Bojong Soang WWTP) facilities (48%) all have access by 2030 (55.2% of health benefits), and (2) people that have access to a well-managed off-site or fecal sludge management system increased from 7% to 73%, (44.6% of health benefits). Associated averted health impact due to irregular flooding events is only US$ 0.3 million.

Convenience and time savings are among the top five reasons for having a latrine in the home area (Winara et al., 2011). Based on Winara et al. (2011) a mean annual gain of US$ 77 million was determined for an additional 45% of the population in 2030 having access to their own latrine facilities. This estimate is conservative as (1) it excludes travel needs for urination purposes, and (2) time is valued conservatively at 30% of the GDP per capita at hourly values.

US$ 13.9 million of the total US$ 23 million reduction in water treatment cost will accrue to the public water utilities and their consumers, while industries are expected to benefit US$ 4.7 million annually. The value of farmed fish yields is expected to be US$ 4 million annually.

The combined environmental benefits (increased land value, reduced dredging, averted maintenance costs of hydro-electric facilities) amount to US$ 17 million, of which nearly 90% is attributed to increases in land value based on annual land sales. The benefits of reduced dredging (even assuming no decomposition or organic waste) has minor benefits.

Table 7.7 shows the estimated reuse benefits based on the per capita production features and resource values (Table A7.4 in Appendix Chapter 7). 87% of the US$ 147 million yearly potential revenues are from municipal solid waste, 11% from domestic wastewater treatment and recovery and reuse of its resources and recycling, and 2% from industrial wastewater treatment and recycling. The potential demand for recoverable resources is higher than the potential supply through recovery (Table 7.7), ranging from a factor 13 for water to a factor 1.6 for plastic.

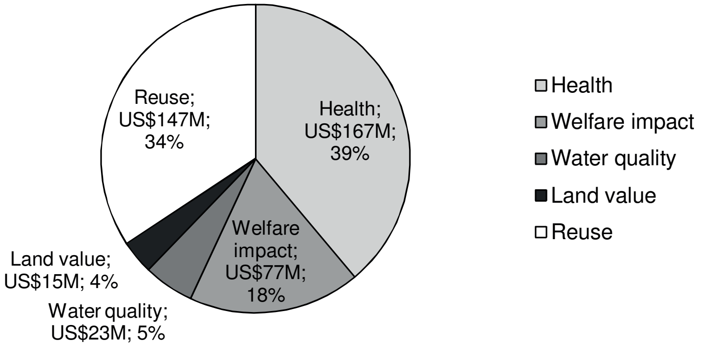

Figure 7.6 Contribution of calculated overall economic benefits expressed in million US$ (total US$ 430 million) of each monetized impact (Scenario 6). Sedimentation (US$2M; 0% contribution) and Dam maintenance (US$0.1; 0% contribution) are not shown

Table 7.7 Resource recovery potential, sector of recovery (Domestic, Industrial or MSW), potential demand, recovery percentage and annual economic values associated with reuse options based on baseline prices (Table A7.4 in Appendix Chapter 7)

Parameter		Recoverable resources per sector and potential demand						Total revenues (million US$/year)
	Domestic WWT	Industrial WWT	MSW	Total recovery	Potential demand	Unit	Recovery percentage	
Compost	91	-	351	442	1,240	kt/y	36%	44.2
Plastic	-	-	228	228	366	kt/y	62%	45.5
Paper	-	-	193	193	1185	kt/y	16%	38.6
Electricity	27	-	89	116	78.8	GWh/y	147%	11.6
Water	-	43	-	43	563	Mm^3/y	8%	2.6
Struvite	4.2	-	-	4.2	35	kt/y	12%	4.1
Total economic value								**146.6**

7.3.6 Assessment of Benefits versus Costs

Following the anticipated benefits (Figure 7.6) and corresponding investment and operational costs (Table A7.6 in Appendix Chapter 7, Section 8) the BCR was calculated (Figure 7.7). The BCR varied between the interventions. The highest BCR of 3.2 is achieved by implementing simple technologies (S3: 50% ST), in other words an economic return of US$ 3.2 is anticipated for each US$ 1 invested. Because of higher costs for AT compared to ST, the BCR is expected to be lower for the AT scenario (BCR in S3: 50% AT = 2.06). The lowest BCR (0.52) is found in scenario 4 (industrial interventions alone). A joint approach tackling both domestic and industrial pollution results in a BCR ranging from 1.83 (S5: 50% AT) to 2.64 (S5: 50% ST). However, simple technologies were not found sufficient to improve the water quality to levels approaching class II, especially in terms of nutrient (N, P) removal (Figure 7.5).

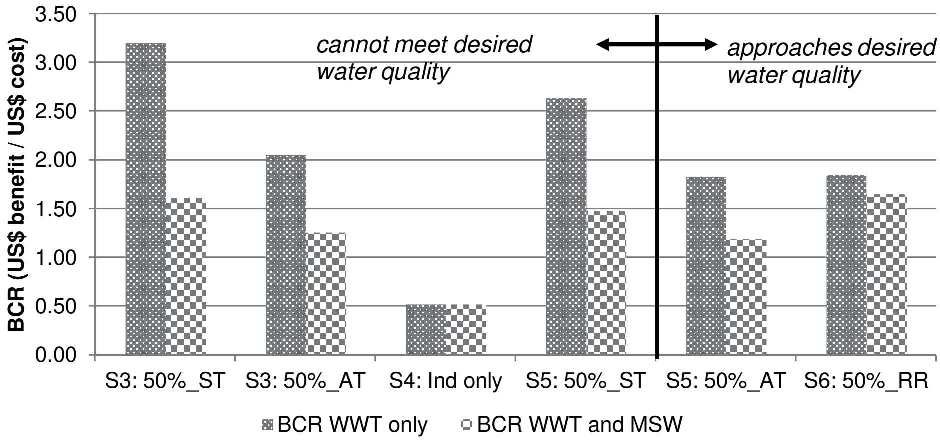

Figure 7.7 Calculated BCR per analyzed scenarios, differentiating the BCR in which only wastewater treatment (WWT) interventions are considered and the BCR that considers both WWT and municipal solid waste (MSW) interventions. Scenarios that approach the target water quality are S5: 50%-AT andS6: 50%-RR

The economic returns on combined wastewater and solid waste interventions are lower than the returns on wastewater interventions only (Figure 7.7). Economic costs related to absence of solid waste services are associated with unhygienic living conditions (Winara et al., 2011), loss of tourism developments or value of land (Alam & Marinova, 2003). Loss of land value, however, contributes to only a fraction (4%) of total related economic impact (Figure 7.6) and on their own do not outweigh the estimated costs (see Table A7.6 in Appendix Chapter 7) to establish the MSW management systems. The willingness of households to pay for solid waste collection and treatment services has been better established compared to wastewater services in Indonesia (WSP, 2011). This may be attributed to direct visibility of improving solid waste management (Winters et al., 2014). Consequently, there is a larger potential for recovering some of the costs through MSW tariffs paid by households compared with tariffs for wastewater services. Potential revenues from fees were excluded from the BCR analysis, but are relevant for development of a cost-effective wastewater and solid waste management system.

Additional benefits of resource recovery from MSW can be a driver for improving water quality. The BCR (including MSW) of scenario 5 (applying AT) is 1.19 and will increase to 1.65 by applying resource recovery (Table 7.8). The BCR of scenario 6 with MSW recovery is even higher than the BCR of Scenario 5 applying ST (1.49) showing that required additional costs to improve the water quality can be financed through the sale of resources recovered from solid waste. However, application of resource recovery from wastewater only results in a minor increase in BCR (from 1.83 to 1.85) compared to applying only advanced technology. Thus, from a financial perspective using existing market prices, the additional investments to recover resources from wastewater outweigh the benefits by a small margin.

In case recovered resources are sold at only half the current market price (Table 7.8) the BCR of resource recovery (S6) is lower than for AT, but still higher than 1. The BCR may change depending on the lifespan of capital stock (Table 7.8). A lifespan of 40 years results in a BCR approaching 5 (S3: 50%_ST). A major part of the cost (Table A7.1 in Appendix Chapter 7) is related to sewer system developments that have typically much longer potential lifespans (even up to 100 years) (Petit-Boix et al., 2014) and therefore it is likely the BCR will be higher than the baseline BCR of 3.2 for that same scenario (S3: 50%_ST).

Table 7.8 Calculated Benefits Costs Ratio (BCR) and five alternative BCR's distinguishing (A) only WWT based BCR or (B) WWT and MSW based BCR

Category	Sub category	A. WWT costs and benefits						B. WWT and MSW costs and benefits					
		S3: 50%_ST	S3: 50%_AT	S4: industrial only	S5: 50%_ST	S5: 50%_AT	S6: 50%_RR	S3: 50%_ST	S3: 50%_AT	S4: Industrial only	S5: 50%_ST	S5: 50%_AT	S6: 50%_RR
BCR	Baseline BCR	3.20	2.06	0.52	2.64	1.83	1.85	1.62	1.26	0.52	1.49	1.19	1.65
	Resource prices 50% of baseline	3.20	2.06	0.52	2.64	1.83	1.79	1.62	1.26	0.52	1.49	1.19	1.37
	Health impact 50% of baseline	2.22	1.42	0.52	1.86	1.29	1.34	1.12	0.87	0.52	1.05	0.84	1.33
	40 year capital lifespan	4.94	3.01	0.60	3.80	2.58	2.61	2.11	1.65	0.60	1.91	1.54	2.18
	15 year capital lifespan	2.60	1.70	0.48	2.19	1.53	1.55	1.40	1.09	0.48	1.29	1.03	1.42
	Access time gained 50% of baseline	2.75	1.76	0.52	2.28	1.58	1.61	1.39	1.08	0.52	1.28	1.03	1.50

7.4 Discussion

7.4.1 Added value of integrated approaches

Evaluating the economic performance of wastewater and solid waste interventions is a complex process, involving many variables and alternative combinations and coverage levels of interventions. Therefore a methodology was developed that combines several assessment methods and data sources in order to support decision making. The added value of the integrated approach allows for a nuanced view on interrelations compared to single cause-effect relations (Mirakyan & De Guio, 2013). Thus the effects of different interventions on water quality, resource recovery potential, and related economic returns could be evaluated in parallel (Figure 7.5, Figure 7.6, and Figure 7.7). This parallel evaluation provides significant benefits in a dynamic context (Pollack, 2009). It also addresses the need for a method that can quantitatively evaluate a set of sanitation alternatives to resolve trade-offs across sustainability dimensions (social, environmental, and economic), as identified by Guest et al. (2009)

7.4.2 Added value of the approach in practical applications

The practical application of the integrated approach is first demonstrated in the analysis of contribution of pollution per sector (industry, domestic or agriculture) related to the pollution prevention costs. The large contribution of domestic pollution was unambiguous and confirmed in a sensitivity analysis (see also Appendix Chapter 7, Section 7). Presented results are in line with findings of Suharyanto & Matsushita (2011) who determined that households contributed 55%, industries 40% and agriculture 6% of BOD pollution entering the Saguling reservoir. Suwarno et al. (2013) demonstrated the importance of fertilizer use management to avoid future coastal eutrophication in Indonesian Rivers, which corresponds with the large nutrient load as a result of agricultural activities (25% for P) determined in the current study. Despite a relative low (28%) contribution of industrial COD pollution, 35% of COD can be reduced by industrial interventions, whereas the investment costs for industrial interventions are less than 10% of the domestic interventions (Table A7.6 in Appendix Chapter 7). Further, the number of industries is only a fraction (~1%) of the number of households in the Citarum area and monitoring interventions would be much more practical than monitoring individual household connections. Thus, although COD pollution from industry is relatively small, it is more cost effective (> factor 5) than domestic, which may help a policy maker in prioritizing interventions.

Secondly, the integrated approach supports determination of cost-effective interventions. The added value of applying more advanced technologies or switching more people to a sewer system showed that required additional investments can be justified from the point of nutrient removal, but less so from COD removal (Figure 7.4). In addition, the use of software tools like RIBASIM to model and estimate the impact of discharged pollution loads on the anticipated water quality allows the policy maker to relate interventions and their cost to applicable water quality standards.

Thirdly, linking the resource recovery potential and its revenues to its potential demand may benefit formulation of policies or increase government involvement to foster financial sustainability of sanitation facilities (Murray & Ray, 2010). The value of recoverable resources from solid waste has resulted in a very active, but informal waste recovery sector in Indonesia (Chaerul et al. 2013; Sasaki & Araki 2013). In addition, the demonstrated potential recovery of resources exceeding the agricultural demand allows for selective marketing, focusing on safe reuse (e.g. on non-edible crops) (WHO, 2006). Electricity production from the joint wastewater and solid waste facilities is potentially higher than the demand for domestic wastewater and supports the potential for a joint development of wastewater and solid waste facilities (Zitomer et al., 2008).

Fourthly, monetizing both direct use and indirect non-use values of sanitation implementation in relation to achievable surface water quality enables the formulation of a cost and environmental effective approach. The performed analysis demonstrated that the most cost effective scenario (S3: 50%_ST) with the highest BCR differs from the scenario reaching the required water quality (e.g. S5: 50%_AT). Therefore, a policy maker needs to prioritize between these two options. As a cost effective strategy, application of advanced technologies may be restricted to the most highly densely populated urban areas (where most pollution is produced). Alternatively, a phased approach in which first simple (low cost) technologies are implemented that are later replaced, converted or extended by systems that allow for nutrient removal (Spiller et al., 2015). Monetizing benefits may further help to raise funds from other sources or actors that benefit from improved water quality, such as residential project developers or tourism sites (Hutton, 2013).

The outcomes of the study were formulated in a planning document for the Indonesian government (Kerstens et al., 2013) and confirms our hypothesis that quantification of tangible outputs using the presented approach can support policy-makers in the field.

7.4.3 Options for extending the approach

The presented framework can be further extended given the following considerations:

- To assess the sustainability of interventions and ensure that pollution is being removed and not displaced, environmental emissions other than water pollution (COD, N, P), such as odor or greenhouse gasses may be included. The effect of greenhouse gasses emitted by low cost technologies (e.g. anaerobic filters or septic tanks) is excluded from the current evaluation;
- In the determination of the water quality, several assumptions were made that may affect obtained results and could be incorporated in a next phase (see also Appendix Chapter 7 Section 9). First, a connection between surface and ground water was assumed in which infiltrated septic tank effluent load directly influences the surface water quality. Second, RIBASIM model disregards biological conversion of pollutants in the surface water, whereas these are observed in the field (Hart et al., 2002). Third, all interventions are assumed to be designed, constructed, operated and maintained correctly, which may be optimistic in view of current practice (De Vries, 2012; ADB, 2013b). Fourth, the effect of dumped solid waste on water quality is excluded. Finally, surface water pollution from animal manure was excluded;
- The low BCR of industrial interventions (0.52, see Table 7.8) and the weak mandatory industrial regulation in Indonesia (D'Hondt, 2013) may suggest limited possibilities to implement industrial pollution prevention. However, alternative means to spur Indonesian industries to comply with environmental standards such as public disclosure (the regular collection and dissemination of information about firms' environmental performance) have been shown to be effective (Blackman, 2010);
- Aerobic technologies were used as industrial references, whereas the use of anaerobic technologies may result in lower investment and/or operational costs (Rajeshwari et al., 2000; Van Lier, 2008);

- Not all economic impacts were quantified in this study (see also Appendix Chapter 7, section 10), such as consumption of fish imbibing toxic wastes or otherwise infected (Lasut et al., 2008), reduced land subsidence and improved recreational values (Day & Mourato, 1998; Alam, 2008). In addition, long-term impacts on the river and population of industrially discharged toxins and heavy metals were excluded and would specifically increase the BCR of scenario S4 (industrial intervention);
- Applying advanced technologies (AT) will improve water quality (Figure 7.5), but will not increase quantified health or welfare impact. At the same, anticipated long-term effects of reduced eutrophication and less impacted ecosystem functioning (Suwarno et al., 2013) were not quantified, whereas these would further increase the BCR;
- The BCR considers the overall societal perspective, whereas different costs and benefits are incurred and enjoyed by different stakeholders. Thus, the costs of domestic interventions are to a large extent paid for by the national and local governments (in Indonesia ~ 70%) and to lesser extent by individual households (Kerstens, et al., 2016), whereas industries typically pay the costs of the interventions themselves (De Vries, 2012). Benefits of improved water quality as a result of interventions can be either increased revenues (e.g. sale of recovered resources) or averted costs (e.g. lower water treatment costs) which benefit a single party, or are generalized to the population (e.g. averted health or time costs) which benefit society as a whole (Alam, 2008). In the elaboration of a planning document, the incidence of costs and benefits should be further detailed.

7.5 Conclusions

In this study, an integrated method was presented that quantifies the economic costs and benefits of wastewater and solid waste interventions in relation to water quality improvements and resource recovery potential. The approach provides added value in the decision making process in a complex and dynamic context since it helps resolve trade-offs across different dimensions of sustainability (e.g. social, environmental and economic).

Identification of pollution sources and the impact of interventions on discharged pollution loads allows for prioritizing of actions. By simultaneously modelling the water quality and cost impact of variations in (1) type of technology and (2) the household numbers switching from poor-performing septic tanks to off-site systems, insight into the cost-effectiveness of environmental policies is provided. This allows a policy maker to optimize economic and water quality benefits.

In the presented case of the Upper Citarum River, domestic interventions applying simple technologies were most attractive, with an estimated BCR of 3.2. However, to achieve the target water quality both industrial and advanced domestic WWT technologies would be required, leading to an estimated BCR of 2.0. Resource recovery from MSW were found to be a driver for improving water quality, as benefits through the sale of recovered resource outweighed the additional costs to improve the water quality.

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7.6 References

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Appendix Chapter 7

Section 1 Description of applied wastewater treatment systems

Kerstens, Leusbrock, et al., (2015) describe three types of wastewater systems: on-site, community based (CBS) and off-site (medium central and centralized) systems. On-site systems (e.g. septic tanks) do not require a sewer system and typically treat black (toilet) water only. Taking the effect of direct discharge pollution of greywater in account, this intervention shows low pollution removal efficiencies per person served (COD ~40%, BOD ~45%, N ~15%, P ~5%) (Kerstens et al. 2015). A community based systems, using a simplified sewer system (Mara & Broome 2008), serves typically 50-100 households. Treatment occurs in a anaerobic baffled system with removal efficiencies of COD ~80%, BOD ~85%, N ~15%, P ~15% and coliforms ~ 2 log (Ulrich et al. 2009). Medium centralized and centralized off-site systems use a simplified or (pumped) sanitary system to collect and transport the wastewater and serve, respectively, up to 5,000 and 50,000 households. Removal efficiencies of off-site system may range from similar values as community based systems (anaerobic filters) to higher efficiencies (COD ~ 90%, BOD ~ 97%, TN~ 90%, TP ~67%, coliforms ~ 3 log) for technologies including enhanced nutrient removal (Kerstens et al. 2015). Following the impact interventions may have on the environment and public health in relation to their costs, the use of on-site system is promoted for low density urban and rural areas only. For higher density rural areas (peri-urban) the use of community based systems is considered, whereas for high density urban areas off-site systems are preferred (Kerstens et al. 2015). Table A7.1 shows applied per capita investment and operational costs for applied technologies.

Table A7.1 Applied per capita investment and net operational costs (OPEX) [a] per Simple, (ST), Advanced (AT) and Resource recovery (RR) technology (Kerstens et al. 2015)

<table>
<tr><th>Parameter</th><th>Type</th><th>Unit</th><th>on-site</th><th>CBS</th><th>Medium Central</th><th>Central</th></tr>
<tr><td>Sewer costs</td><td></td><td rowspan="4">US$/cap</td><td>0</td><td>114</td><td>229</td><td>324</td></tr>
<tr><td rowspan="3">Treatment + land</td><td>ST</td><td rowspan="3">100</td><td rowspan="3">104</td><td>60</td><td>95</td></tr>
<tr><td>AT</td><td>188</td><td>118</td></tr>
<tr><td>RR</td><td>194</td><td>118</td></tr>
<tr><td rowspan="3">Net OPEX</td><td>ST</td><td rowspan="3">US$/cap/year</td><td rowspan="3">2.1</td><td rowspan="3">2.7</td><td>5.1</td><td>6.8</td></tr>
<tr><td>AT</td><td>10.5</td><td>7.8</td></tr>
<tr><td>RR</td><td>6.3</td><td>3.2</td></tr>
</table>

[a] In the Net OPEX costs all operational costs (labour, energy, chemicals, sludge disposal) as well as revenues from recoverable resources (energy, struvite and compost) at their anticipated selling prices (Table A7.4) were included

Section 2 Maps of study area

Figure A7.1 and Figure A7.2 show maps of the study area

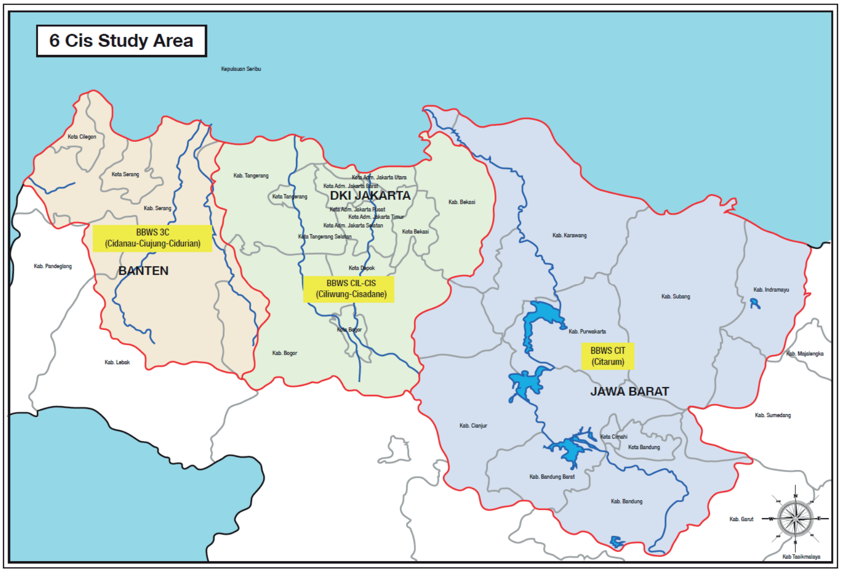

Figure A7.1 Location of the BBWS CIT (Citarum Greater Basin Territory Centre *"Balai Besar Wilayah Sungai")* in Indonesia (MoPW 2011)

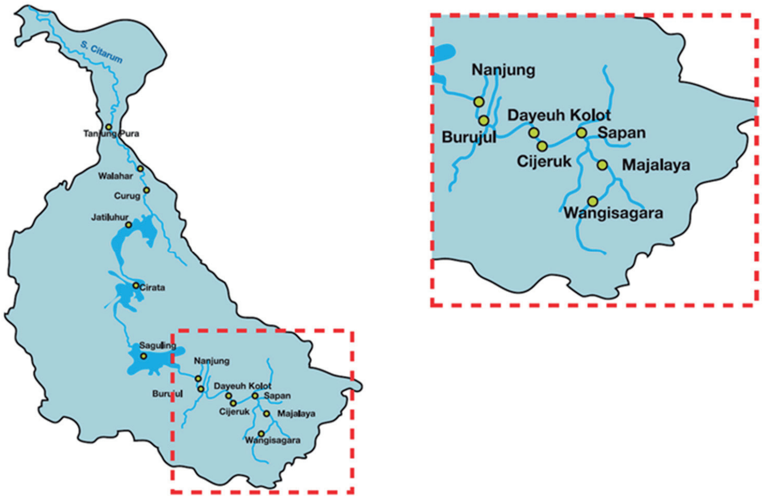

Figure A7.2 Location of the Upper Citarum River basin within the Citarum basin (Kerstens et al. 2013)

Section 3 Description of applied industrial WWT

To determine the type and costs of industrial interventions, three types of wastewater treatment plant (WWTP) designs, for each typical scale were prepared (Kerstens et al. 2013). Designs were based on:

1. Textile industry producing Batik (Section 3.1);
2. Textile industry producing other types of textile (no reactive dyes) (Section 3.2);
3. General industrial wastewater treatment plant (Food/Beverage and paper pulp) (Section 3.3).

In case effluent is reused treatment occurs in a system as described in Section 3.4.

For each of these types of "uniform" WWTP construction, CAPEX, OPEX and total running costs have been determined, based on quotation of international suppliers in Indonesia (Nijhuis, Aqua), and project visits to industries in the area (Frisian Flag and Ultra Jaya) and contractors and author's estimate (Table A7.2).

Table A7.2 Determined Investment and operational costs as a function of treated daily flow for three types of industrial wastewaters, based on MPS & Nijhuis (2012)

	Investment (US$/m^3/d treated)			OPEX (US$/m^3 treated)		
Flow (m^3/d)	Batik textile	non-batik textile	other industries	Batik textile	non-batik textile	other industries
0-100	1890	2556	2556	0.47	0.57	0.55
100-500	1368	1686	1698	0.37	0.44	0.41
500-1000	1114	1145	1234	0.35	0.39	0.36
1000-2000	907	848	908	0.32	0.35	0.34
> 2000	700	581	667	0.30	0.32	0.31

For cost determination it is assumed that 50% of the textile industries are batik industries (applying system 1) industry and 50% produces a different type of textile. In addition it is assumed that 50% of all industries that already have a treatment system need to upgrade their system before 2030. The brief overview of each design is shown below.

Section 3.1 Textile industry using reactive dyes (Batik industry)

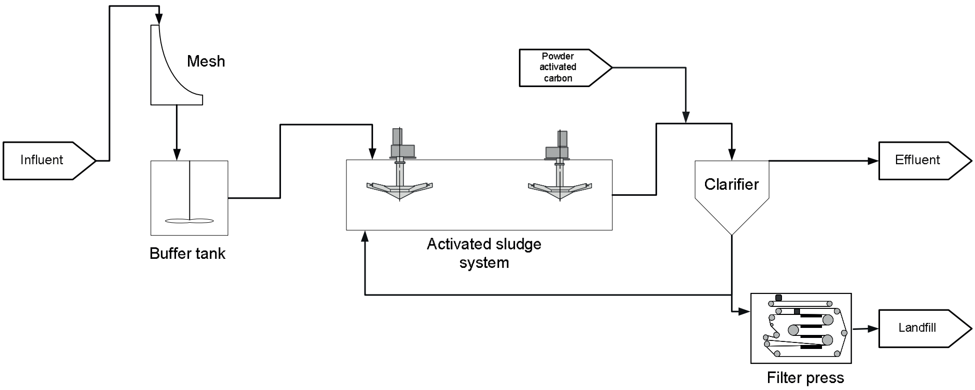

Figure A7.3 Process Flow Diagram of a WWTP for a textile industry using reactive dyes

Influent first passes a mesh to remove suspended solids (e.g. pieces of cloth) after which the influent is discharged to a mixed buffer tank (Figure A7.3). The buffer tank aims to stabilize variations in both quantity and composition of influent. Possible pH control can take place in this tank. After the buffer the wastewater is fed to an activated sludge system, where biomass (bacteria) convert organic pollutants (COD, BOD) into CO_2 and new biomass using oxygen that is brought in by aeration equipment (surface or bubble). Also nitrogen components are removed through nitrification and denitrification. Phosphorous in the wastewater is incorporated in the cell mass after which it is removed from the system via the waste activated sludge. Optionally iron or aluminum salts can be dosed to enhance phosphorus removal (chemical precipitation). The treated effluent and sludge are separated in the clarifier. To remove the reactive dyes (color), activated carbon is dosed prior to the clarifier. The treated effluent leaves the clarifier from the top, whereas the sludge is returned to the activated sludge tank. Because some sludge is produced during the process the excess sludge needs to be removed from the system and, after thickening and dewatering is disposed in a landfill.

Section 3.2 System 2: Textile industry using non-reactive dyes

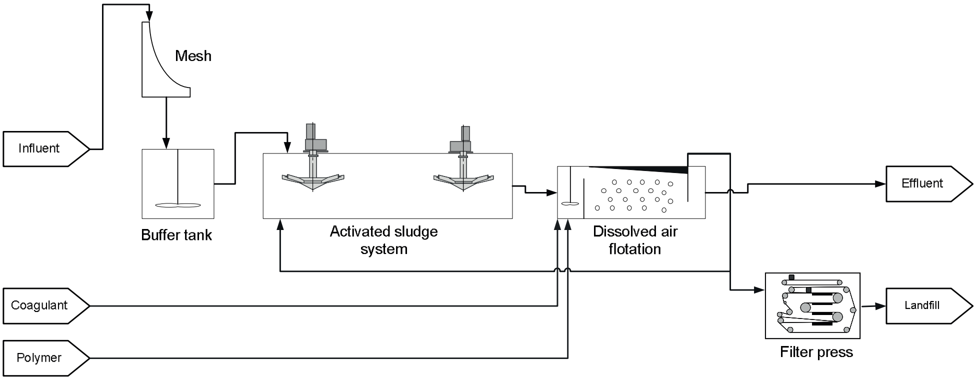

Figure A7.4 Process Flow Diagram of a WWTP for a textile industry using non-reactive dyes

Similar to system 1 Influent first passes a mesh to remove small scale solid particles after which it is introduced in a mixed buffer tank (Figure A7.4). After the buffer the water wastewater is introduced in an activated sludge system. Unlike system 1, in system 2 color (dye) removal and separation of sludge from the treated effluent happens in one DAF (Dissolved Air Flotation) unit. In this unit, dissolved organic dyes as well as a considerable part of phosphorus are "glued" together using coagulation and further turned into flocs (together with the sludge) using polymers. The produced flocs adsorb to introduced air that floats from the bottom to the top. The floating layer is continuously removed from the top using a skimmer. Similar to system excess sludge needs to be removed from the system and, after thickening and dewatering, disposed off in a landfill.

Section 3.3 System 3: Other types of industry

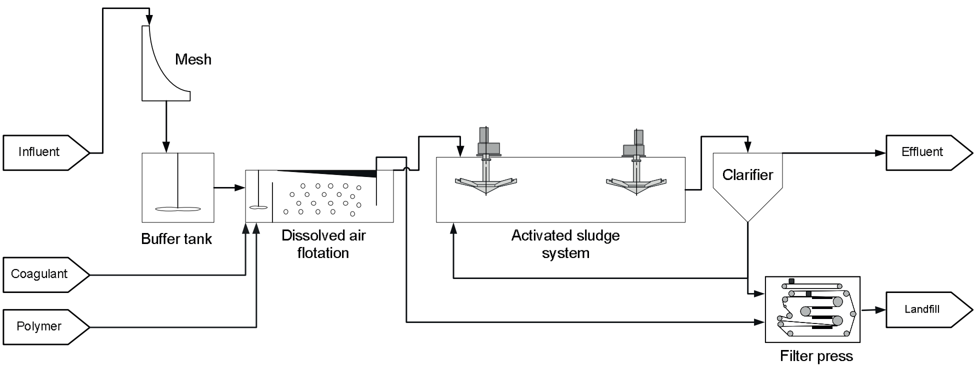

Figure A7.5 Process Flow Diagram of a WWTP for other types of industry

Similar to system 1 and 2 Influent first passes a mesh and a mixed buffer tank (Figure A7.4). To remove Fat Oil and Grease (typical for dairy industry) or non-biodegradable solids (typical for paper and pulp processes) pre-treatment takes place in a DAF (Dissolved Air Flotation) unit. In which, similar to system 2, through addition of coagulant and flocculants a considerable part of organic pollutants (and phosphorous) is removed prior to treatment in the activated sludge system (removal of organic and nitrogen). The treated effluent and sludge are separated in the clarifier. The treated effluent leaves the clarifier from the top. Sludge is returned to the activated sludge tank and partly extracted and send to a landfill.

Section 3.4 Post treatment of industrial effluent

In case effluent is reused treatment occurs in a system as presented in.

In order to reuse effluent, it needs to be further treated on (1) particles, (2) organic (dissolved) components, (3) high salt concentration and pathogens. To achieve this a multi barrier system set-up is proposed (Figure A7.6), consisting of:

(1) Sand filtration: the effluent of the WWTP still contains some levels of suspended solids (typical 10-20 mg/l), which are removed in a sand filter. Typically coagulant Al^{3+} salts are added to improve this step;

(2) After biological treatments, there may be still organic components that could not be degraded biologically, but can be removed in an AOP (Advanced Oxidation Process) in which a chemical reagent (e.g. NaOCl; Sodium hypochlorite) is used to oxidize the components;

(3) All remaining organic components will then pass an activated carbon filter in which these components are adsorbed to the filter material;

(4) after removing all organic components and particles, the water passes a reverse osmosis (R.O.) unit, in which all salts will to a large extent be removed;

(5) Because in an R.O. salts are removed, addition of minerals (e.g. Na^+, Cl^-) may be required and these will be added again in a mineralization step;

(6) finally the water will pass a disinfection step (e.g. UV, ozone) to assure that no pathogens/viruses are present.

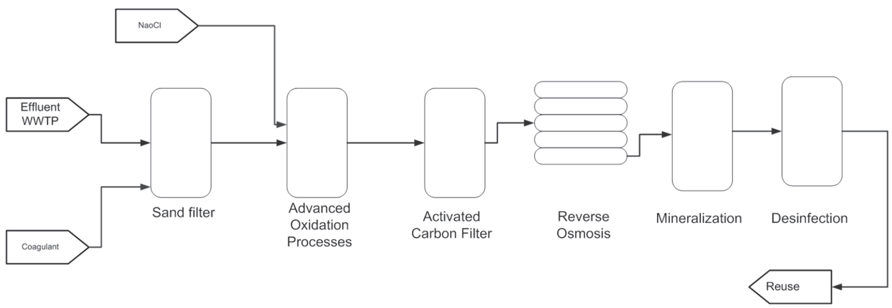

Figure A7.6 Process Flow Diagram of post-treatment system for industrial effluent

Section 4 Description of applied Municipal Solid Waste systems

Municipal Solid Waste (MSW) system selection is based on residential features (Table 7.4 of Chapter 7) and distinguishes conventional system and systems applying *Reduction*, *Reuse*, *Recycling* (3R) of waste. In low density rural areas promotion of home composting is applied (Mehta & Movik 2010). For higher density populated rural and urban areas digestion and composting of organic waste and recovery of plastic and paper is considered. Applied costs are shown in Table A7.3.

Table A7.3 Applied MSW Capital Expenditures (CAPEX) and net Operational Expenditures[a] (OPEX) (Kerstens et al. 2015) [a]**.**

Parameter	Unit	Conventional		3R home	3R decentral		Centralized 3R	
		Rural	Urban	Rural	Rural	Urban	Compost	Digest & Compost
CAPEX	US$/cap	45.2	72.9	2.6	73.3	95.3	57.0	69.8
Net OPEX	US$/cap/y	8.7	10.2	-3.7	-3.4	-4.5	-3.8	-4.4

[a] In the Net OPEX costs, the anticipated recoverable resources and their anticipated selling prices (Table A7.4) were included

Per capita resource recovery potential and economic values (Table A7.4) were based on Kerstens, Leusbrock, et al. (2015) and Aprilia et al. (2012).

Table A7.4 Per capita recoverable resource production rates from Domestic WWT and Municipal Solid Waste (MSW) and their economic value

Sector	Component	Per capita production[a]	Economic value
Domestic WWT	Struvite (as a source for P)	0.82 kg struvite/cap/year	975 US$/t[a]
	Electricity production	5.35 kWh/cap/year	0.1 US$/kWh[a]
	Compost from on-site	4.4 kg/cap/year	100 US$/t[b]
	Compost from off-site	15 kg/cap/year	100 US$/t[b]
MSW	Compost	39 kg/cap/year	100 US$/t[b]
	Electricity production	18 kWh/cap/year	0.1 US$/kWh[a]
	Plastic recovery	26 kg/cap/year	2000 US$/t[b]
	Paper recovery	22 kg/cap/year	2000 US$/t[b]

[a] based on Kerstens, Leusbrock, et al. (2015); [b] based on Aprilia et al. (2012)

Section 5 RIBASIM description

RIBASIM is a generic model package for simulating the behaviour of river basins under various hydrological conditions, was used to simulate the effect of different interventions on water quality development in the Upper Citarum River (Deltares 2009; Gonenc et al. 2014). It links the hydrological water inputs at various locations with the specific water-users in the basin. RIBASIM enables the user to evaluate a variety of measures related to infrastructure, operational and demand management and to see the results in terms of water quantity, water quality and flow composition. The Upper Citarum River basin has nine catchment areas in which water abstraction by domestic, industrial and agriculture activities and catchment of rain (based on 20 years rainfall sequence) were modelled (MoPW 2011; Kerstens et al. 2013). Based on pollution loads produced in each catchment area and resulting water flows concentrations are calculated. Figure A7.7 and Figure A7.8 show the RIBASIM overlay and model.

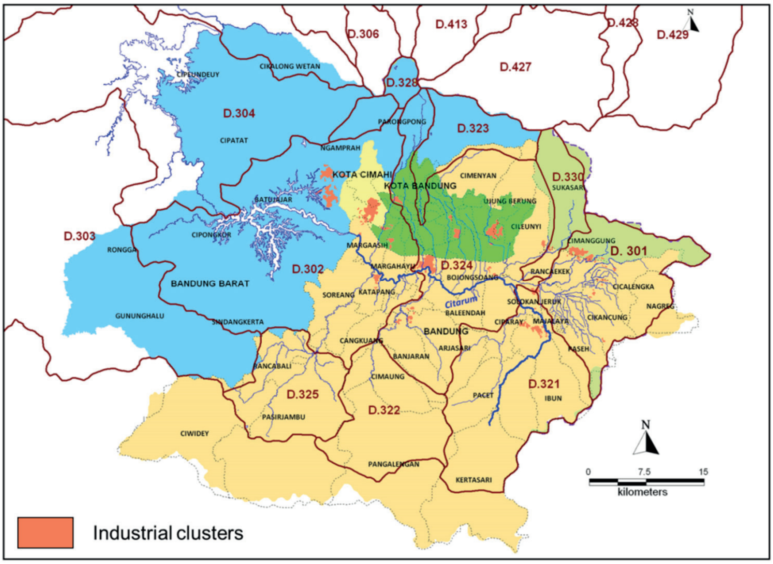

Figure A7.7 Prepared overlay of catchment areas on the administrative maps of the Upper Citarum River (Kerstens et al. 2013)

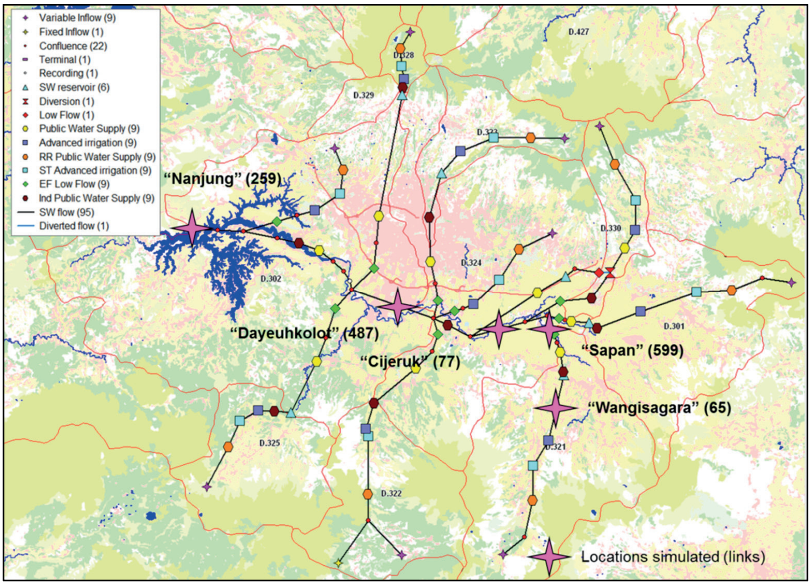

Figure A7.8 Prepared RIBASIM model in this study and approximate locations and corresponding links in RIBASIM (Kerstens et al. 2013)

Section 6 Calculated relative contribution of pollution by actor

The contribution of domestic, industrial and agricultural sources differs with (1) the parameter (COD, BOD, TN and TP), (2) the expected domestic pollution coefficient and (3) the performance industries. The highest variation in contributions is obtained combining a high pollution coefficient of domestic sources (high domestic load) with the best case industries (low industrial load) and vice versa (Figure A7.9). The reference domestic COD contribution is 66%, with maximum values of 77% (high pollution coefficient and best case performing industries) and minimum 44% (low pollution coefficient and worst case performing industries). BOD pollution follows the pattern of COD pollution, but TN and TP loads show a relative high contribution of about 20% from agricultural activities.

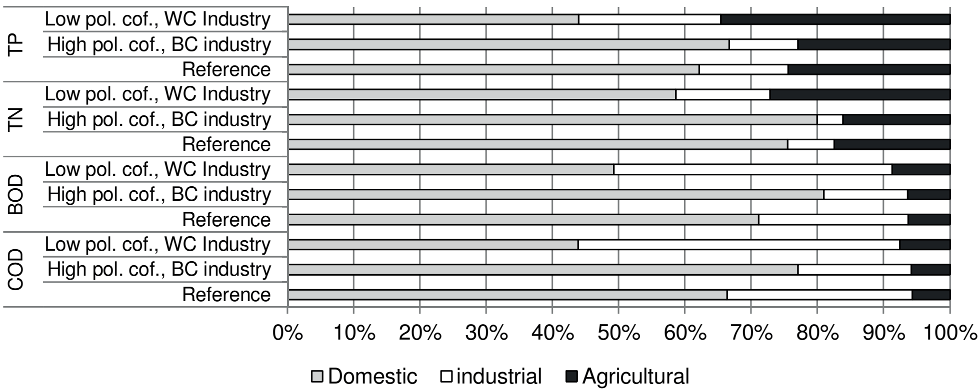

Figure A7.9 Contribution of Domestic, Industrial and Agricultural sources per parameter (COD, BOD, TN and TP) for baseline scenario and scenarios with highest, lowest variations in contribution by varying domestic pollution coefficient (pol. cof.) and industrial worst case (WC) and best case (BC) cases

Section 7 Pollution loads discharged with different interventions

Table A7.5 shows the effect of described interventions on calculated pollution discharged to the Upper Citarum River by sector (domestic, industrial, agricultural). It shows that in case the 2010 (baseline) access values to wastewater facilities are not increased by the year 2030 (no additional interventions) the COD discharged will increase from 585 (S1) to 722 t/d (S2).

Addressing only industries (Scenario 4 in Table A7.5) reduced the total COD pollution load from 722 (scenario 2) to 565 ton COD/d, equivalent to a 22% reduction. Combining domestic and industrial interventions (S5: 75% AT) results in a removal of COD (67%), BOD (73%), TN (50%) and TP (32%) compared to S2.

the relation between domestic pollutants removed and domestic investment costs (Figure 7.5 in Chapter 7) demonstrates that additional investments in advanced technologies, compared to simple technology can be justified from the point of nutrient removal, but less so from COD removal.. The total investments for S3: 25% ST were US$ 1.15 billion to remove 260 t COD/d and 8.7 t N/d, corresponding with 4 million US$/t COD/d and 120 million US$/t N/d. For S3: 75% AT investments were nearly US$ 2 billion to remove 327 t COD/d and 51 t N/d, corresponding with US$ 5.8 million per t COD /d and US$ 37 million per t N/d.

Table A7.5 Total pollution loads per defined scenario

parameter	sector	2010[a]	2030																			
		S1: Baseline	S2: No intervention	S3: 25% ST	S3: 25% AT	S3: 50% ST	S3: 50% AT	S3: 75% ST	S3: 75% AT	S4: Ind only	S5: 25% ST	S5: 25% AT	S5: 50% ST	S5: 50% AT	S5: 75% ST	S5: 75% AT	S6: 25% ST	S6: 25% AT	S6: 50% ST	S6: 50% AT	S6: 75% ST	S6: 75% AT
COD (t/d)	DM	388	490	228	205	210	184	193	163	490	228	205	210	184	193	163	228	205	210	184	193	163
	Ind	163	194	194	194	194	194	194	194	37	37	37	37	37	37	37	29	29	29	29	29	29
	Agr	34	38	38	38	38	38	38	38	38	38	38	38	38	38	38	38	38	38	38	38	38
	Total	**585**	**722**	**459**	**436**	**442**	**416**	**424**	**395**	**565**	**302**	**280**	**285**	**259**	**267**	**238**	**295**	**272**	**277**	**251**	**260**	**230**
BOD (t/d)	DM	188	238	97	81	88	69	79	58	238	97	81	88	69	79	58	97	81	88	69	79	58
	Ind	60	70	70	70	70	70	70	70	12	12	12	12	12	12	12	10	10	10	10	10	10
	Agr	17	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19
	Total	**264**	**327**	**186**	**170**	**177**	**159**	**168**	**148**	**269**	**128**	**112**	**119**	**100**	**110**	**89**	**126**	**110**	**117**	**99**	**108**	**87**
TN (t/d)	DM	68	86	77	44	77	39	76	35	86	77	44	77	39	76	35	77	44	77	39	76	35
	Ind	6	8	8	8	8	8	8	8	3	3	3	3	3	3	3	2	2	2	2	2	2
	Agr	16	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18
	Total	**91**	**111**	**102**	**70**	**102**	**65**	**102**	**60**	**106**	**98**	**65**	**97**	**60**	**97**	**55**	**97**	**64**	**96**	**59**	**96**	**54**
TP (t/d)	DM	12	15	15	10	15	9	15	9	15	15	10	15	9	15	9	15	10	15	9	15	9
	Ind	3	3	3	3	3	3	3	3	2	2	2	2	2	2	2	1	1	1	1	1	1
	Agr	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
	Total	**20**	**24**	**24**	**18**	**24**	**18**	**24**	**17**	**23**	**23**	**18**	**23**	**17**	**23**	**16**	**22**	**17**	**22**	**16**	**22**	**15**

S1: 2010: Baseline situation; **S2**: 2030: Reference case; same WWT access percentage as 2010 applied; **S3**: 2030 100% DM access, varying between simple (ST) and advanced technologies (AT) and switch factor (25%-50% and 75%); **S4**: 2030: Industrial WWT intervention; 100% of big (> 1,000 m^3/d), 90% of medium (500-1,000 m^3/d), 80% small (100-500 m^3/d), and 75% of very small (< 100 m^3/d) sized industries apply intervention. DM WWT follows S2; **S5** Combination of S3 and S4; **S6**: Same as S5, but using reuse recovery technologies for DM, Ind and MSW

Section 8 Determined costs of interventions

Table A7.6 shows the total costs of interventions per scenario (applying a 50% switch). The S2 intervention investments amount to US$ 123 million and relate to covering population growth while maintaining the same access values as in 2010. The US$ 120 million for industrial WWT interventions (S4) are less than 10% of the domestic ST interventions investments. The costs (US$ 396 million) for conventional MSW management (S3 and S5) are 22% (compared to AT) and 30% (compared to ST) of domestic WWT interventions. OPEX are higher for MSW (90 million US$/ton) than for WWT (36 or 59 million US$/ton), and largely related to solid waste collection activities (Kerstens et al. 2015). All intervention costs in the reuse scenario (S6) have increased and total costs increase from US$ 2.3 billion to US$ 2.6 billion. However, as a result of the sale of resources the OPEX decreases from US$ 166 to 25 million per year. In the determination of the BCR, the cost of scenario 2 to compensate for population growth were deducted from costs in scenarios 3, 5 and 6.

Table A7.6 Costs of interventions for described scenarios

	Investment (million US$)				OPEX (million US$/year)			
Scenario	domestic WWT	Industrial WWT	MSW	Total	domestic WWT	Industrial WWT	MSW	Total
S2: No intervention	109	0	13	123	11	0	26	37
S3: Domestic ST 50%	1,299	0	396	1,695	36	0	90	126
S3: Domestic AT 50%	1,781	0	396	2,177	59	0	90	150
S4: Industrial only	109	120	13	243	11	17	26	54
S5: Domestic ST 50%+ Industrial + MSW	1,299	120	396	1,815	36	17	90	143
S5: Domestic AT 50%+ Industrial + MSW	1,781	120	396	2,297	59	17	90	166
S6: 50% RR	1,803	222	579	2,637	37	23	-35	25

Section 9 Considerations for improving the water quality modelling

In our approach to determine the water quality an open connection between surface and ground water was assumed in which infiltrated septic tank effluent load directly influences the surface water quality. Degradation, conversion, sorption that take place in the soil matrix or aquifer may result in a lower load reaching the final surface water and thus result in lower pollution concentrations (ARGOSS 2001; McDowell et al. 2005). Secondly, RIBASIM model disregards biological degradation, conversion and sedimentation of pollutants in the surface water. Because the hydraulic retention time is only a few days (MoPW 2011), these processes may be neglected. However, the Saguling reservoir has a retention time exceeding 80 days and abundance of nutrients has resulted in excessive plant growth (Hart et al. 2002) indicating that biological processes have an impact. Thirdly, all interventions (industrial and domestic) are assumed to be designed, constructed, operated and maintained correctly, which may be optimistic in view of current practice (De Vries 2012; ADB 2013). Fourthly, the effect of dumped solid waste on water quality (as COD, BOD, N and P) is excluded. Despite low solid waste collection and disposal rates in West Java (43% in urban; 3% in rural areas), most of the unmanaged waste is informally incinerated (39% urban and 63% rural) (Ministry of Health 2010), while only a limited amount is discharged into the river (11% urban, 17% rural) (Ministry of Health 2010), which may justify this exclusion. Finally, surface water pollution from animal manure was excluded. In the study area, over 70% of cow manure is collected in stables and composted or digested and applied on (horticultural) land (Parikesit et al. 2005). A further analysis of livestock management could provide additional tools to improve water quality.

Section 10 Economic impacts not quantified in this study

A number of health impacts were not quantified in this study, such as consuming fish that are raised in or exposed to polluted water from municipal and industrial discharges (Alabaster 1986) (Lasut et al. 2008), especially the impacts of mercury on pregnant women (National Research Council 2000) (Rasmussen et al. 2005). These would further increase the benefit of the proposed measures if included.

Reduced land subsidence is not quantified in this study, but is a direct cause of excessive ground water extraction by industries, the municipality, farmers and households. Reported land subsidence rates amount to 7 cm/year and an estimated 1.1 million people live in a flood prone area of the Citarum River basin (Deltares et al. 2012). A study for Java estimated the economic damages of flood damages at US$ 700 million per year (ADB 2011). Based on the proportion of population at risk in the study area, this corresponds with US$ 80 million flood damages. If industries were to use their treated wastewater effluent instead of using ground water (scenario 6), land subsidence rate can be decreased.

Other potentially significant benefits of improved river water quality, though not quantified in the current study, include the pleasures of walking, relaxing and enjoying scenery of local residents and tourists, and letting children play in or around the river as well as boating on the river (Day & Mourato 1998; Alam 2008).

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Chapter 8

Sanitation Nationwide Planning Framework: Synthesis, concluding remarks and outlook

8.1 Motivation

Worldwide, 2.5 billion people lack access to wastewater or solid waste (WHO & UNICEF, 2015).The absence of sanitation impacts public health (Shuval, 2003), the environment (Hart et al., 2002), the economy, people's welfare (Hutton, 2013), and is a lost opportunity for potential resource recovery from wastewater and solid waste (Lettinga, 2006). Including resource recovery in sanitation planning allows for a circular resource management and may become a driver for economic growth (McDonough & Braungart, 2010; Agudelo-Vera et al., 2011). Resource recovery may thus respond to profound changes in the world's population (United Nations, 2014), impacting food security and finite natural resource availability (Cordell et al., 2011).

8.1.1 Current status of sanitation planning and resource recovery in developing countries

The backlog in sanitation development has been partly attributed to the absence of a functional sanitation planning framework (WHO & UNICEF, 2014). Sanitation planning requires integration of health (Malekpour et al., 2013), technical (Larsen et al., 2009), environmental (Suwarno et al., 2013), financial (Ward, 2012), institutional (Kvarnström & Mcconville, 2007), and socio-economical elements (Winters et al., 2014) as well as insight in the demand for sanitation by-products (Murray & Ray, 2010a). To evaluate a set of alternative sanitation systems, a framework is required to resolve trade-offs between costs and benefits across spatial and temporal scales and to assess sustainability dimensions (Guest et al., 2009). In **Chapter 2** (Kerstens, De Mes, et al., 2009), we demonstrated for a Chinese residential area development that an integrated wastewater and solid waste management scheme using resource recovery can result in both environmental and economic benefits, as compared to a conventional baseline scenario. Yet, in many developing countries, sanitation implementation focuses on a single system or sanitation sub-sector, for example either wastewater or solid waste. In Indonesia, for instance, decentralized (communal) wastewater treatment systems (DEWATS) are promoted as the core of the sanitation improvement (Eales et al., 2013).

Thus, in **Chapter 3** (Kerstens et al., 2012) we first evaluated the technical and financial-economic aspects and users' involvement of DEWATS. It was shown that systems generally comply with applicable legislation, yet financial and operational management is often lacking. In addition, we identified the need to link planning and implementation of DEWATS to the residential features in a city wide sanitation strategy.

However, no integrated framework exists in scientific literature that links and quantifies costs and benefits of sanitation intervention with (1) public health and economic improvements, (2) reducing pollution discharge loads and improving water quality, and (3) recovering and applying resources on a national scale. Available sanitation planning frameworks were not found applicable for nationwide planning, since these did not include all population groups or both wastewater and solid waste treatment and resource recovery systems. In addition, available frameworks relied on input parameters that were not readily available on a nationwide scale. Moreover, these frameworks did not include integration with land use planning activities or allowed for identification and budget allocation of implementing institutions. Further, an integrated technical and financial sanitation system analysis under different residential conditions was lacking. This, despite the availability of technical and financial system selection criteria for sanitation systems (Achillas et al., 2013; Tilley et al., 2014) and the demonstrated link between residential features and occurrence of health and environmental issues caused by the absence of sanitation (Wright et al., 2013). Finally, an analysis of resource flows (Brunner & Rechberger, 2004) and insight in potential product markets are required to quantify the potential demand and supply of sanitation products. While a diversity of methods to forecast resource demand have been described (Lyneis, 2000; Janssen et al., 2005; Tilman et al., 2011), there is no framework that combines recoverable resources from wastewater and solid waste that also allows for a nationwide temporal and spatial demand forecast.

8.1.2 Need for a sanitation nationwide planning framework

In this thesis, we developed a framework that can quantitatively evaluate a set of alternative wastewater and solid waste systems. This framework resolves costs and benefits across spatial and temporal scales and includes sustainability dimensions (social, environmental and economic). It was hypothesized that access to sanitation in developing countries can be accelerated by an increased benefit cost ratio resulting from resource recovery. To test our hypothesis, we have used Indonesia as case.

8.2 Sanitation Nationwide Planning framework (SaNaP)

The Sanitation Nationwide Planning framework (SaNaP) distinguishes 3 elements: (1) sanitation system selection, (2) sanitation planning framework, and (3) potential resource demand analysis. These three elements can, in turn, be linked to come to a fourth element to quantify benefits and costs of sanitation interventions.

8.2.1 Sanitation system selection

Chapter 4 provides a technical and financial feasibility analysis of a range of wastewater treatment (WWT) and municipal solid waste (MSW) systems to guide system selection for implementation in different residential settings. Analyzed systems comprised on-site, community-based systems (CBS) and ten off-site WWT systems as well as conventional, centralized and decentralized 3R (**R**educe **R**euse **R**ecycle) MSW systems. Figure 8.1 shows a schematic presentation of this approach, in which a conventional (upper part) intervention is compared with a resource recovery (lower part) based intervention for two different residential features (urban/rural). Depending on the type of area (urban/rural) and selected system, CAPEX and OPEX (together forming the total life cycle costs) are determined.

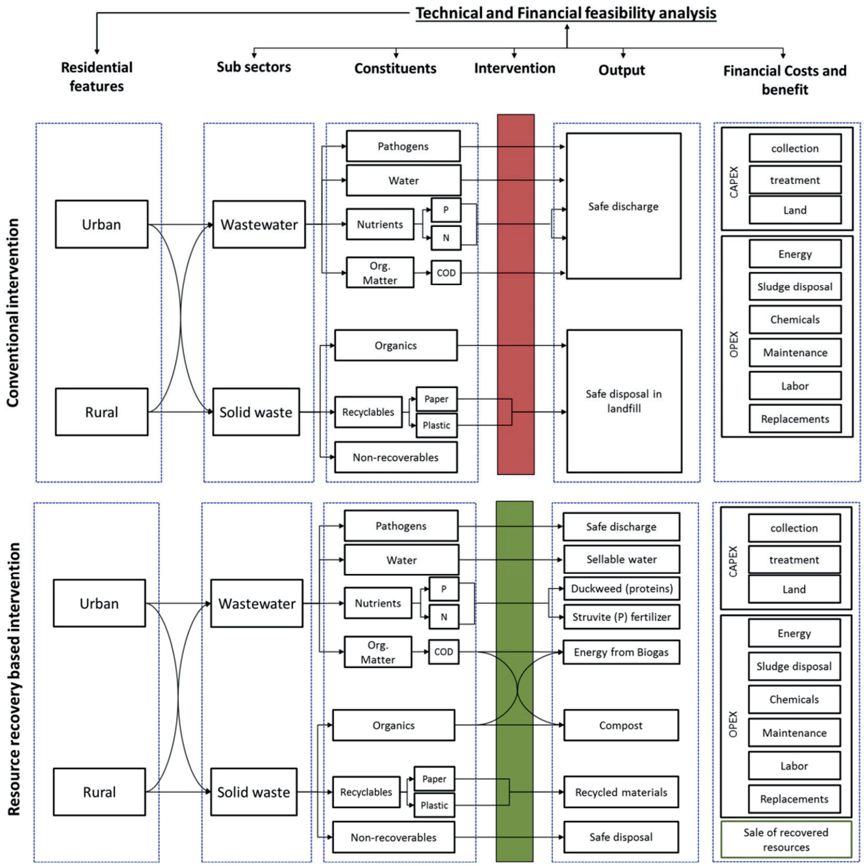

Figure 8.1 Schematic presentation of the approach for determining the technical and financial feasibility of wastewater and solid waste technologies and the link to residential features

8.2.2 Sanitation planning framework

In **Chapter 5** a comprehensive framework was developed that directly links a government policy to a nationwide, long-term planning and budgeting for wastewater and solid waste interventions. Figure 8.2 shows how, on the basis of anticipated population development (urban rural, poor and non-poor), sanitation (wastewater and solid waste) systems are selected. Based on government sanitation planning targets, three quantifiable outputs are generated: (1) visualization of required implementation using a geographic information system (GIS), (2) required budgets for people served and number of systems to be constructed, and (3) distribution of budgets for implementation over identified responsible institutions.

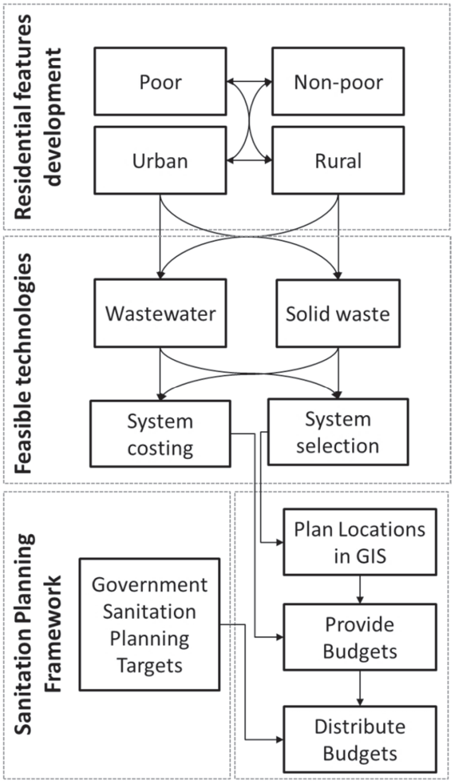

Figure 8.2 Schematic presentation of the framework for assessing planning outputs on the basis of residential developments, feasible sanitation systems and government targets

8.2.3 Quantification of potential demand and supply of sanitation products

Chapter 6 describes the future demand of recoverable resources, based on past consumption trends and future forecast for a selected number of recoverable resources. Figure 8.3 shows how demand for P-fertilizers and compost used for food, horticultural and plantation crop production as well as duckweed demand for aquaculture are determined following production forecasts. Furthermore, future demand for plastic and paper, to substitute conventionally manufactured products, is determined by extrapolation based on past production and consumption patterns. This potential resource demand forecast is then compared to the potential forecasted resource supply from wastewater and solid waste flows.

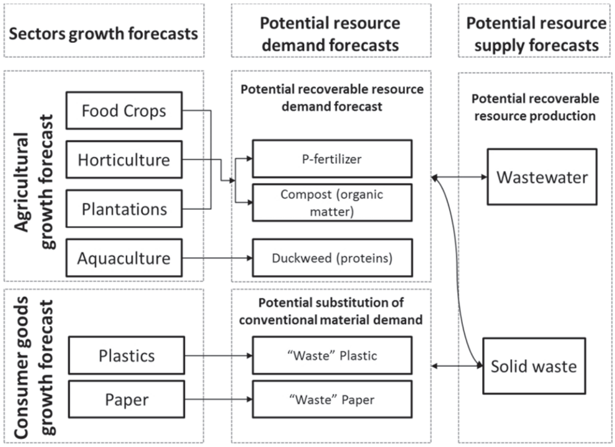

Figure 8.3 Schematic presentation of the framework for forecasting the recoverable resource demand compared to resource recovery from wastewater and solid waste (see Chapter 6)

8.2.4 Quantification of costs and benefits of alternative sanitation interventions

In **Chapter 7**, the three introduced elements were combined in a framework that allows for a quantifiable analysis of benefits and costs of a range of domestic and industrial wastewater and solid waste interventions. The applied framework is presented in Figure 8.4 and builds upon the previous three elements. Three quantifiable values are generated: (1) water quality improvements as a result of selected WWT systems, (2) potential resource recovery from WWT and MSW systems compared to the potential demand (see Figure 8.3) and, (3) the Benefit Cost Ratio (BCR) in which monetized benefits (health, access time, improved water sources & environment, land values and sale of recovered resources) were compared to required costs of interventions (CAPEX and OPEX).

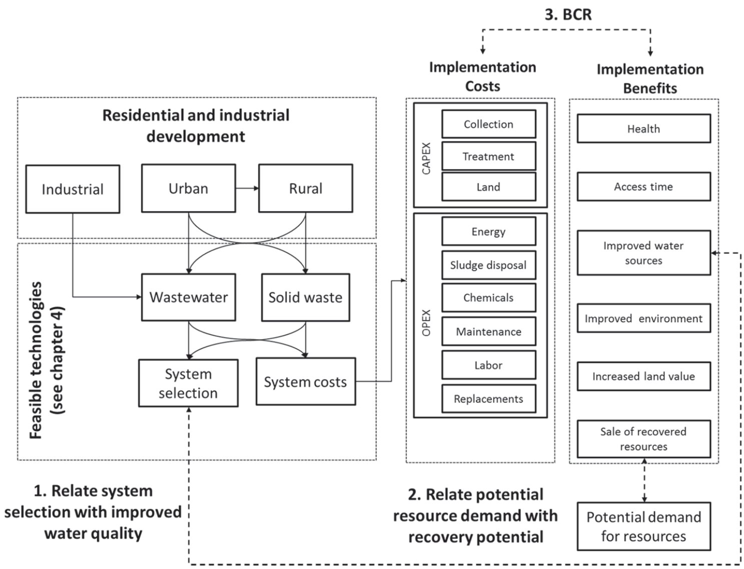

Figure 8.4 Schematic presentation of the methodology used to (1) relate water quality to applied WWT systems, (2) relate the potential resource recovery to the potential demand and (3) determine the Benefit Cost Ratio (BCR)

8.3 Synthesis of the integrated SaNaP framework

Applying the nationwide population development data (of Indonesia) to the combined three elements (sections 8.2.1 - 8.2.3), allows for verification of the hypothesis that resource recovery can accelerate access to sanitation in developing countries.

Moreover, it enables a quantitative evaluation of alternative WWT and MSW systems to resolve costs and benefits across spatial and temporal scales in relation to system costs, pollution loads, production and consumption parameters of resources (e.g. energy, area, sludge), and potential resource demand and supply.

First, Figure 8.5 shows how the technical and financial feasibility analysis (Chapter 4) allows for a WWT and MSW system selection and their associated CAPEX (US$/cap) and OPEX (US$/cap/year) based on residential area criteria. Applying these residential dependent criteria on the anticipated Indonesian residential developments in combination with the "government sanitation planning target" enables us to determine the number of people that must be served and the corresponding number of WWT and MSW systems that must be implemented (see also Chapter 5). Once the specific costs of all described WWT[1] and MSW systems have been determined, the total CAPEX (US$), OPEX (US$/year), and total lifecycle costs (US$ per 20

[1] *One on-site, one community based and ten off-site systems: (1) Anaerobic Filter, (2) Aerated lagoon, (3) UASB (Upflow Anaerobic Sludge Blanket)-DW (Duckweed)-RBC (Rotating Bio Contactor), (4) Conventional Activated Sludge (CAS), (5) CAS with enhanced N&P, (6) Aerobic Granular Sludge (AGS), (7) MBR (Membrane Bioreactor), (8) CAS + Resource Recovery (3R), (9) AGS + 3R and (10) MBR + 3R. On-site (septic tank) and community based systems (anaerobic baffled reactor + anaerobic filter) were not varied.*

years) for all described systems can be compared using the planned implementation path. The impact of selected systems on the cost can be evaluated for spatial (national, provincial and city wide) and temporal scales (period 2015-2035).

Second, Figure 8.5 shows the total pollution loads discharged (COD, BOD, N and P) applying different implemented WWT off-site technologies. Discharged pollution loads are presented in time and space, combining the planned sanitation development and the specific per capita wastewater pollution discharge (derived in Chapter 4). The framework thus gives a quantitative evaluation of the impact of system selection on the environment.

Third, Figure 8.5 shows how the total, system dependent, resource production and consumption (e.g. energy, sludge produced, recovered phosphorus/ paper/plastic and land required) can be determined on the same temporal and spatial scales. Information on required resources (e.g. energy, chemicals or land) to implement and run a facility can be used to evaluate the applicability of a selected system in the regional context and its development (e.g. the planned provision of electricity to a certain location after a number of years).

Finally, Figure 8.5 shows a comparison between spatial and temporal potential resource demand (Chapter 6) on the one hand and production of recoverable resources on the other hand. Comparing the potential resource supply with its demand enables an assessment of the potential to close material cycles in a certain location and time.

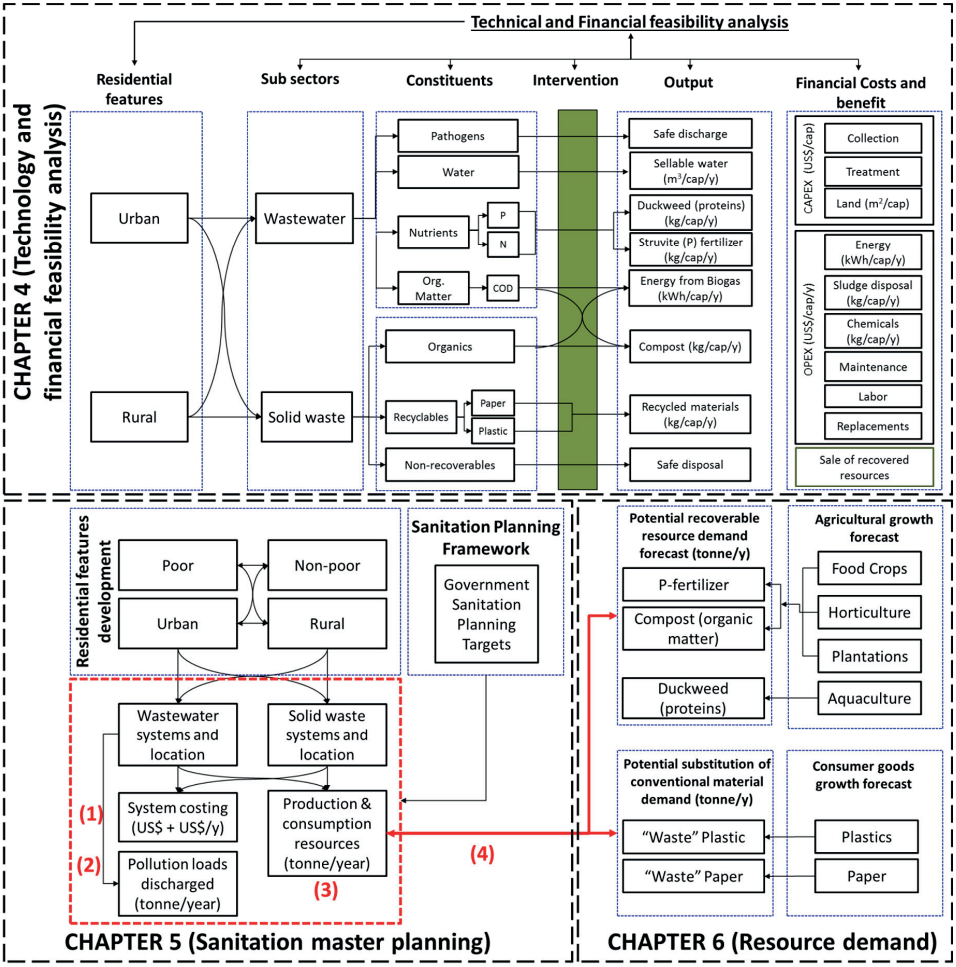

Figure 8.5 Sanitation National Planning (SaNaP) framework in which Chapter 4 (technology and financial feasibility analysis), Chapter 5 (sanitation master planning) and Chapter 6 (potential resource demand) are combined. The four numbered items show the spatial and temporal determination of (1) system costs, (2) wastewater pollution discharged, (3) production and consumption parameters, (4) potential resource demand and recovery

8.4 Results of nationwide application of SaNap for Indonesia

To support policy makers in the evaluation of different sanitation systems, quantification of potential benefits (e.g. health, environment) and associated implementation costs is required (Ward, 2012). Resolving these trade-offs with the different available implementation alternatives is a complex and dynamic matter which is guided by SaNaP. The potential of SaNaP to evaluate system costs, pollution loads, production and consumption parameters and potential resource demand and supply for a set of alternatives is shown for the case of Indonesia.

8.4.1 Spatial and temporal WWT and MSW system costs evaluation

The WWT and MSW costs determined in Chapter 4 are, among others, dependent on land and resource (e.g. electricity) prices. Since land and resource prices vary from one place to the other and in time (Navastara & Navitas, 2012), an impact analysis of these variations can support system selection. Figure 8.6 and Figure 8.7 show a comparison of the total cumulative costs (CAPEX and OPEX and Total Lifecycle Costs (TLC) of on-site CBS and off-site interventions for a land price of 500 US$/m2 (Figure 8.6) and 100 US$/$m^2$ (Figure 8.7) for Indonesia at the end of the planning period (2035). The center top of the radar chart diagram shows the total CAPEX, OPEX and TLC presented of planned on-site, CBS and anaerobic filter (An. Fil.) technologies as the sole off-site treatment technologies. Moving clockwise, the next axes show the same relative costs in case Conventional Activated Sludge (CAS) systems were applied. The Total Life Cycle Costs are presented as Net Present Values (NPV) as explained in Chapter 4 and reflect the mutual differences of only the off-site treatment technologies and their land costs. They exclude TLC of on-site systems and CBS as well as sewer systems, since these costs are the same for all technologies.

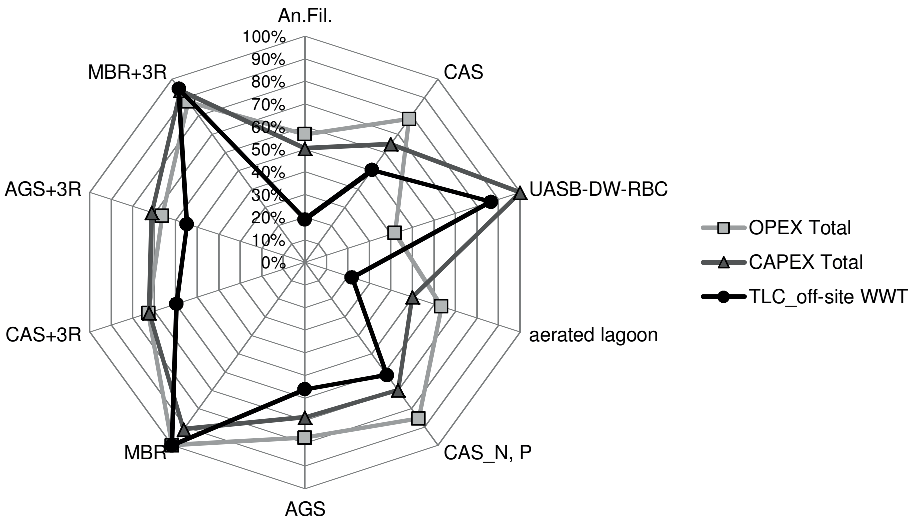

Figure 8.6 Relative OPEX, CAPEX and NPV of interventions for Indonesia (2035) applying the indicated off-site technology (An Fil, CAS until MBR + 3R) as the sole technology for central and medium centralized treatment plants with a land costs of 500 US$/$m^2$. OPEX and CAPEX include all CAPEX and OPEX (on-site, CBS and off-site systems), whereas TLC only includes the treatment + land costs of off-site technologies. The 100% value corresponds with the highest determined value, whereas the 0% value corresponds with an actual zero value

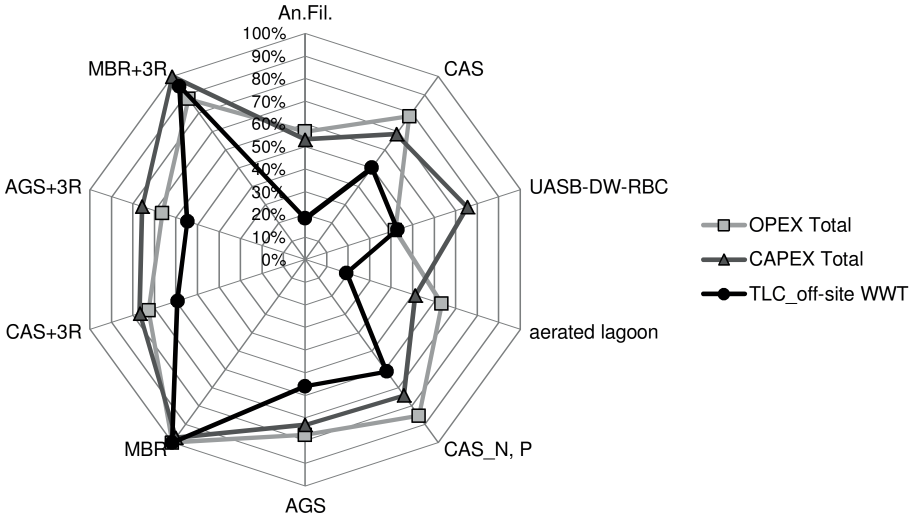

Figure 8.7 Relative OPEX, CAPEX and NPV of interventions for Indonesia (2035) applying the indicated off-site technology (An Fil, CAS until MBR + 3R) as the sole technology for central and medium centralized treatment plants with a land costs of 100 US$/m^2

Figure 8.6 and Figure 8.7 show that land prices particularly impact the CAPEX and total lifecycle costs (as NPV) of land intensive treatment technologies. For example, in areas with high land prices (Figure 8.6; 500 US$/m^2) the land intensive UASB-Duckweed pond–RBC shows the highest CAPEX and nearly the highest TLC (NPV) of all technologies. In contrast, at lower prices (Figure 8.7; 100 US$/m^2) the TLC decreases from nearly 90% to about 40% compared to the highest TLC (100%) of the MBR. It shows that land intensive treatment technologies are not feasible in land scarce (urban) areas. In case the anaerobic filter technology was selected as the sole off-site system, the CAPEX (100 US$/m^2, land price) would be about half of that of the highest costing (MBR) technology. The TLC of the anaerobic filter is only 20% of that of the MBR. Additional resource recovery for off-site technologies (CAS/AGS/MBR + 3R) results in a lower TLC compared to these same technologies without resource recovery, albeit a marginally (only a few percentage points) difference. This shows that application of resource recovery technologies results in a more financially favorable situation, while producing an equal effluent quality (see also Chapter 4). For example, on a nationwide scale over a 20 year period, the TLC of Aerobic Granular Sludge (AGS) system with resource recovery (compost, struvite and energy) is about half a billion US$ less than when resource recovery is not applied using the AGS.

Insight in cost development of alternative MSW management scenarios can support the decision making process on the introduction of resource recovery systems. For example, Figure 8.8 (Indonesia; 2035) shows the comparison of MSW system costs, using different levels of collected waste processed applying 3R. The level of 3R applied in this evaluation are (1) 0% 3R; (2) selected 3R % (31%) to reach a reduction of 20% solid waste being landfilled (see Chapter 4), and (3) all (100%) of solid waste is treated using 3R. Despite about 25% higher investment costs, the TLC of systems applying 3R is more favourable due to lower OPEX. Only 10% of these reduced operational costs can be attributed to reduced collection and transport costs, as a result of decentralized processing. The largest gain is in the decreased treatment and disposal costs and is attributable to the sale of recovered resources (compost, plastics and paper). In the case of 0% 3R scenario, the cumulative disposal and 3R OPEX amounts to US$ 805 million over 20 years (100%) and minus US$ million 260 (= profit) for 100% 3R, indicated as -35% in Figure 8.8. The reduction of land required for combined landfill and centralized 3R activities outside the residential areas (Figure 8.9) is 20% for the 31% 3R scenario and 62% for the 100% 3R scenario compared to a conventional system (0% 3R). At the same time, the level of land required in residential areas (typically more expensive) for decentral 3R stations is higher for the scenarios in which 3R is applied. Land availability was identified as a barrier for development of communal 3R systems (Aprilia et al., 2012) and for that reason not favored in high density urban areas (Chapters 4 and 5).

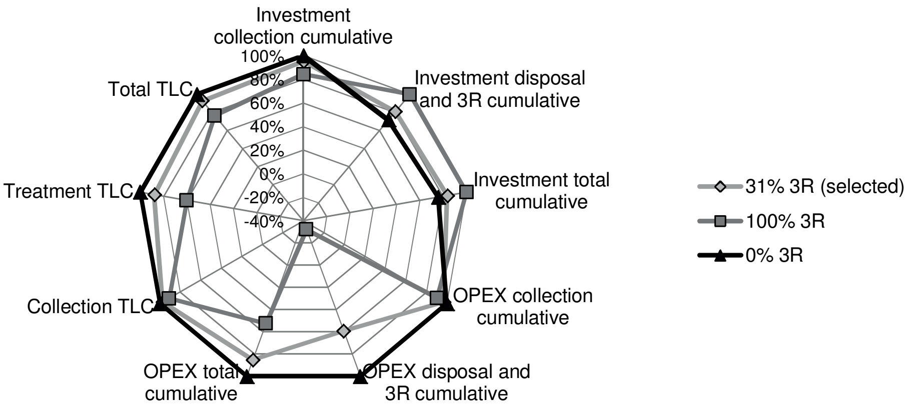

Figure 8.8 Indonesia (2035): Comparison of OPEX, CAPEX and NPV of interventions applying three levels (0%, 31% and 100%) of waste processing using 3R. OPEX, CAPEX and TLC distinguish collection, treatment and total costs. Negative costs correspond with net profit resulting from the sale of recovered resources and are found when comparing OPEX of the 100% 3R with the 0% 3R Scenario

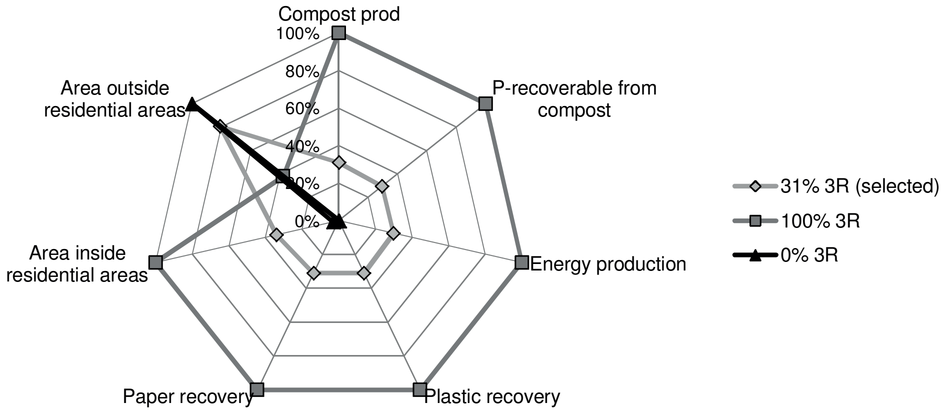

Figure 8.9 Indonesia (2035): Comparison of resource production and consumption parameters of interventions applying three levels (0%, 31% and 100%) of waste processing using 3R

8.4.2 Impact of different WWT technologies on discharged pollution loads

Evaluation based on costs aspects only favoured the application of low cost, non-nutrient removing anaerobic filter technology (Figure 8.6 and Figure 8.7). However, evaluation on the basis of discharged pollution loads, such as phosphorus (Figure 8.10A, B) and to a lesser extent COD (Figure 8.11) do not favour the anaerobic filter. The difference in the cumulative COD load discharged by on-site, CBS systems and any of the off-site technologies (Figure 8.11) is minor (<10%), since only 35% of the population is planned to be connected to off-site systems. The difference in COD removal efficiencies of off-site technologies is only about 10% (Chapter 4). The calculated P-load discharge in 2020 in Indonesia as a whole shows limited differences in the P-removed (~10%) between the selected technologies (Figure 8.10). Despite a planned access of 100% by 2020, less than 15% of the population is connected to an off-site system (see Chapter 5), while the remaining population is connected to a septic tank or community based system with poor (5%) phosphorus removal efficiencies. Therefore, the added value of applying nutrient removal technologies with associated higher costs may seem marginal.

However, when looking at the most urbanized province (Jakarta DKI) in 2035, the differences in P discharge between the evaluated technologies is considerable (Figure 8.10). By 2035, 60% of the population is planned to be connected to an off-site WWTP, based on the 50% switch factor from current on-site users to off-site users as explained in Chapter 5. In contrast, 40% will continue to use septic tanks. Technologies with high P-removal (e.g. MBR, AGS) show a reduction of P (diamond shapes in Figure 8.10) to the environment of 40% compared to the use of anaerobic filter systems. The application of the UASB-DW-RBC would result in an even higher reduction (~45%), but this system is hardly feasible in Jakarta setting due to its large footprint, since it is more than 100 times that of technologies such as the AGS or MBR (Chapter 4). This comparison shows the relevance of evaluating sanitation interventions and their pollution discharge or resource recovery on a temporal and spatial scale. Overall, it confirms that, in sanitation planning, there is no “one size fits all” solution (Guest et al., 2009).

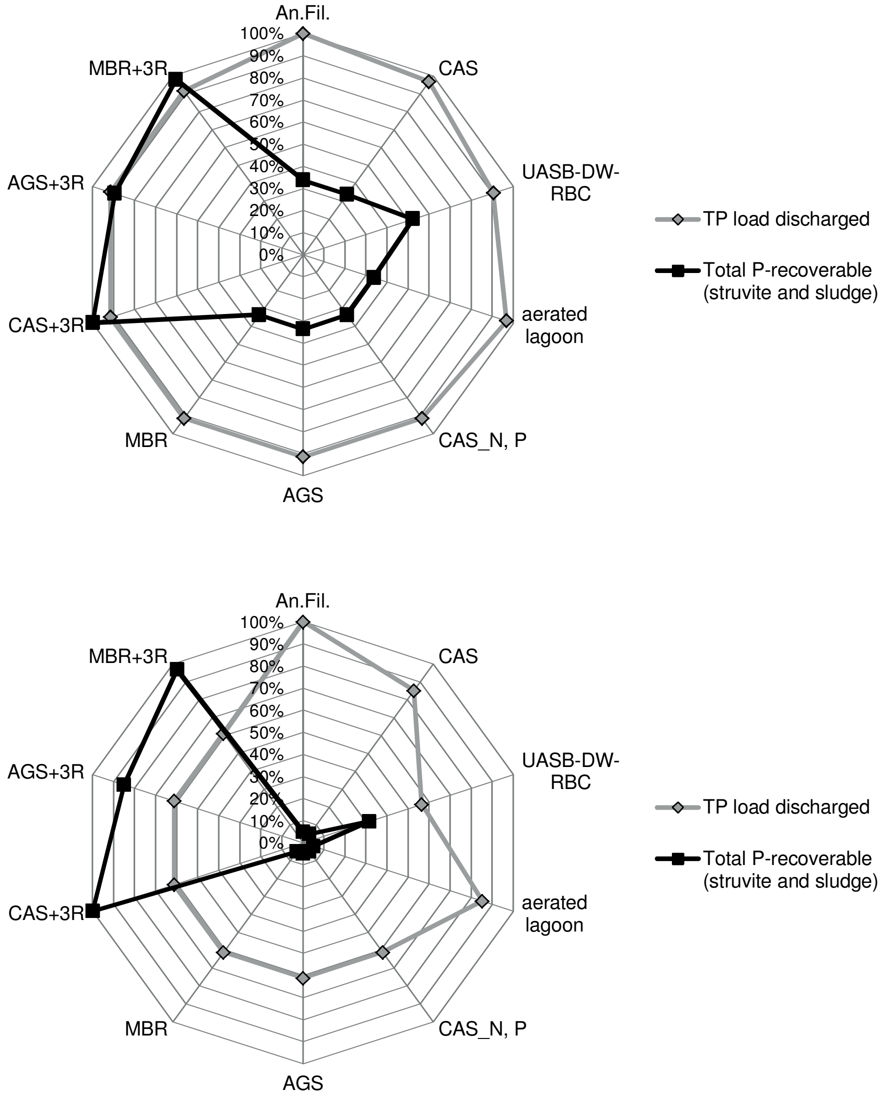

Figure 8.10 (A upper Figure, B lower Figure) Cumulative (on-site, CBS and indicated off-site systems combined) Total phosphorous (TP) load discharged and Total P recoverable (as struvite composted sludge and duckweed). Figure **A**: Indonesia (2020); Figure **B**: DKI-Jakarta (2035)

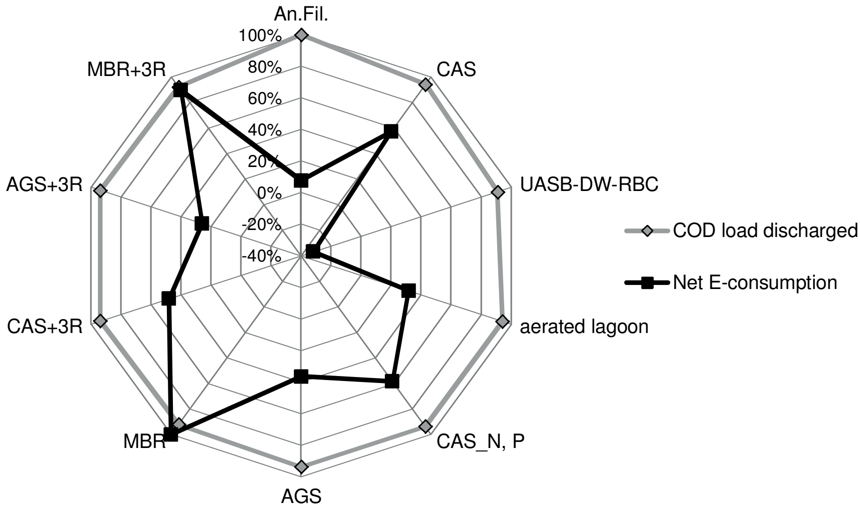

Figure 8.11 Indonesia (2035): Cumulative (on-site, CBS and indicated off-site systems) COD load discharged and net energy production or consumption. A negative value corresponds with a net energy producing system and is found for the UASB-DW-RBC system

8.4.3 Spatial and temporal evaluation of production and consumption resources

SaNaP enables a spatial and temporal evaluation of WWT and MSW resource production and consumption parameters. Figure 8.10, for example, shows the cumulative recoverable P (either as struvite, compost or in duckweed) using the on-site, CBS and indicated off-site technologies. When no additional resource recovery is applied for the off-site technologies (anaerobic filters, CAS, aerated lagoons, CAS, N, P, AGS and MBR), the recovered P is resulting from the processed sludge from septic tanks, applied on-site (see Chapter 4). Comparing the P-recoverable with different off-site technologies between Figure 8.10 A (Indonesia; 2020) and B (Jakarta; 2035), the former shows a relative smaller difference (approximately 65% compared to over 90% maximum for the latter) between these technologies. This is the result of the relatively high (potential) contribution of P-recovered from septic tank sludge compared to the DKI Jakarta case, where far fewer people (40% compared to over 85% nationwide in 2020) apply septic tanks.

Application of the UASB-DW-RBC system results in net energy production (Figure 8.11; square shapes), whereas the MBR requires most energy (see also Chapter 4). Energy dependency should be an important criteria in the selection of off-site systems, especially in areas that are frequently subject to power cuts or have limited access to power (Lettinga, 2006).

Visualizing the absolute amount of recoverable P from wastewater on a provincial level using different off-site technologies may help policy makers or private parties to identify the potential amount of P that can be sold to farmers (Chapter 4). Figure 8.12 shows the province based recoverable P by 2035 and suggests that the highest potential lies in highly populated and urbanized areas located on *Java Barat* (West Java) and *Java Timur* (East Java).

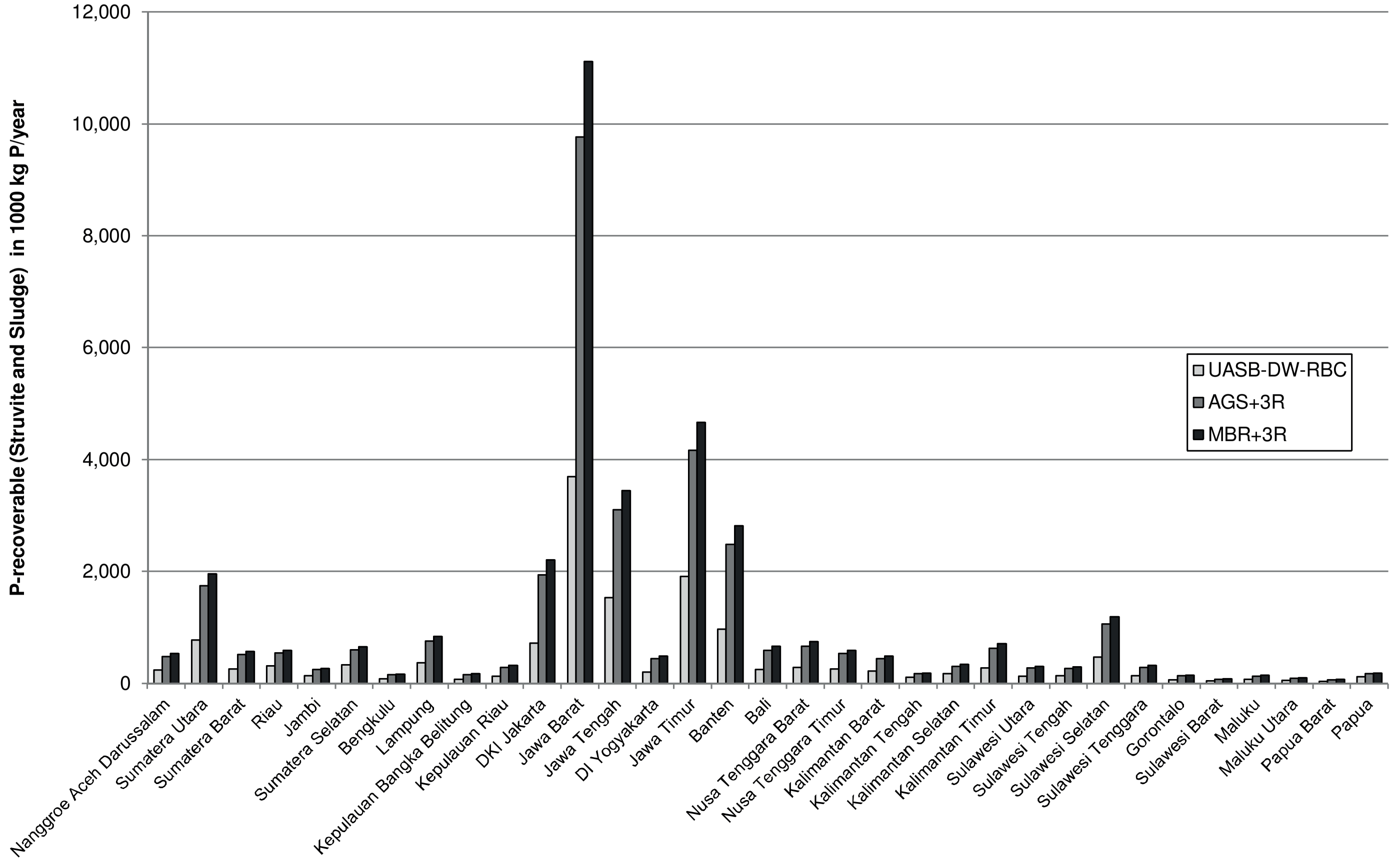

Figure 8.12 Province based (2035): P-recoverable (Struvite and sludge, duckweed) comparing three different systems (UASB-DW-RBC, AGS + 3R and MBR + 3R)

Figure 8.13 shows the time bound (period 2020 and 2035) and nationwide anticipated resource (compost, energy, plastic and paper) recovery potential using the selected (31%) 3R scenario (Chapter 5) and the 100% 3R (in which 3R is applied on all collected waste). It shows that the amount of recoverable resources increases as a result of (1) more people being served by the waste management system in a country with increasing population (relation 2020-2035) and (2) changing government targets with respect to the solid waste reduction targets (relation 31% 3R and 100% 3R). Depending on the potential demand for recoverable resources in an area, the government may set its own ambition level of 3R to both suit the market and stimulate sustainability.

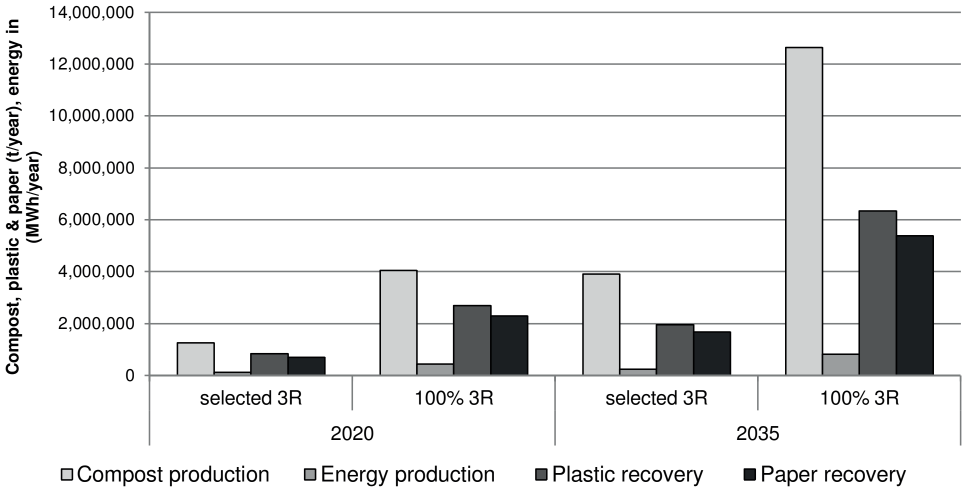

Figure 8.13 Indonesia (2020 and 2035) compost and energy production and plastic and paper recovery from solid waste, applying two levels (31% and 100%) of waste processing using 3R

8.4.4 Spatial and temporal comparison of resource demand and supply

Finally, SaNaP allows for a spatial and temporal evaluation of the potential recoverable resource demand (Chapter 6) in comparison to potential resource recovery following the planned implementation (Chapter 5) and associated resource recovery potential (Chapter 4). Figure 8.14 (phosphorus), Figure 8.15 (compost) and Figure 8.16 (paper) show the cumulative recovery potential from wastewater on-site, CBS and off-site systems (in this example assuming the AGS + 3R) and solid waste (100% 3R) in relation to its demand for each province in the year.

The total (nationwide) P-recovery from wastewater and solid waste in 2035 is nearly 100,000 tonnes P/year. This is almost 10% of the total calculated actual P demand by 2035 (see Chapter 6 for definition of actual). About 20% could be recovered as an inorganic struvite fertilizer, whereas the remaining 80% is recoverable as an organic P-containing compost (from septic and sewage sludge and organic waste processing). Jakarta produces more phosphorus than is required for agricultural purposes (Figure 8.14), which is due to the high recovery potential in combination with the limited agricultural activities in Indonesia's capital. On the scale of Java, more than 30% of the P requirement can be provided through resource recovery, while Java accounts for nearly 60% of the total population (BPS, 2013). In contrast to this, for Sumatra and Kalimantan only 5% could be provided. This low value is related to the (palm oil) plantations that require nearly 60% of all P in the nation (Chapter 6), while the population amounts to less than 30% of people in Indonesia. Less than half of that population is urban, which is the population considered for off-site technologies for selected enhanced P-recovery technologies (Chapter 4). In 2013 the reported inorganic fertilizer P-consumption in Indonesia amounted to 340,000 tonnes of inorganic P, of which nearly 80% is produced (mined) in Indonesia. The export was reported to be 25,000 tonnes P/year (FAO, 2014a). In 2035, the potential recoverable struvite (inorganic) fertilizer from wastewater is 80% of the current (2013) export P-production. Although different time scales are compared here (2013 and 2035), this comparison suggests that the export of recoverable inorganic P from wastewater could considerably strengthen Indonesia's position as a P-exporting country.

Figure 8.15 shows that the degree to which produced compost can fulfil the actual compost demand has less variation on a provincial level than phosphorus, because compost is produced both in urban and rural areas (Chapter 4). In the more urbanized areas (e.g. Java), compost is largely produced in centralized systems. In contrast, in rural areas (e.g. Papua) compost is mainly produced in decentralized systems and on household levels through home-composting of organic waste.

Both phosphorus and compost demand exceed potential recovery from wastewater and solid waste. Since the application of recovered sanitation products is not always perceived as a safe or desired product (Koné et al., 2007; Starkl et al., 2010), it can be considered to selectively market resources, focusing on safe use first (WHO, 2006). Thus, the presented temporal and spatial evaluations of potential supply and demand can help a policy maker to determine what type of marketing and socialization or campaigning (e.g. to farmers) is required.

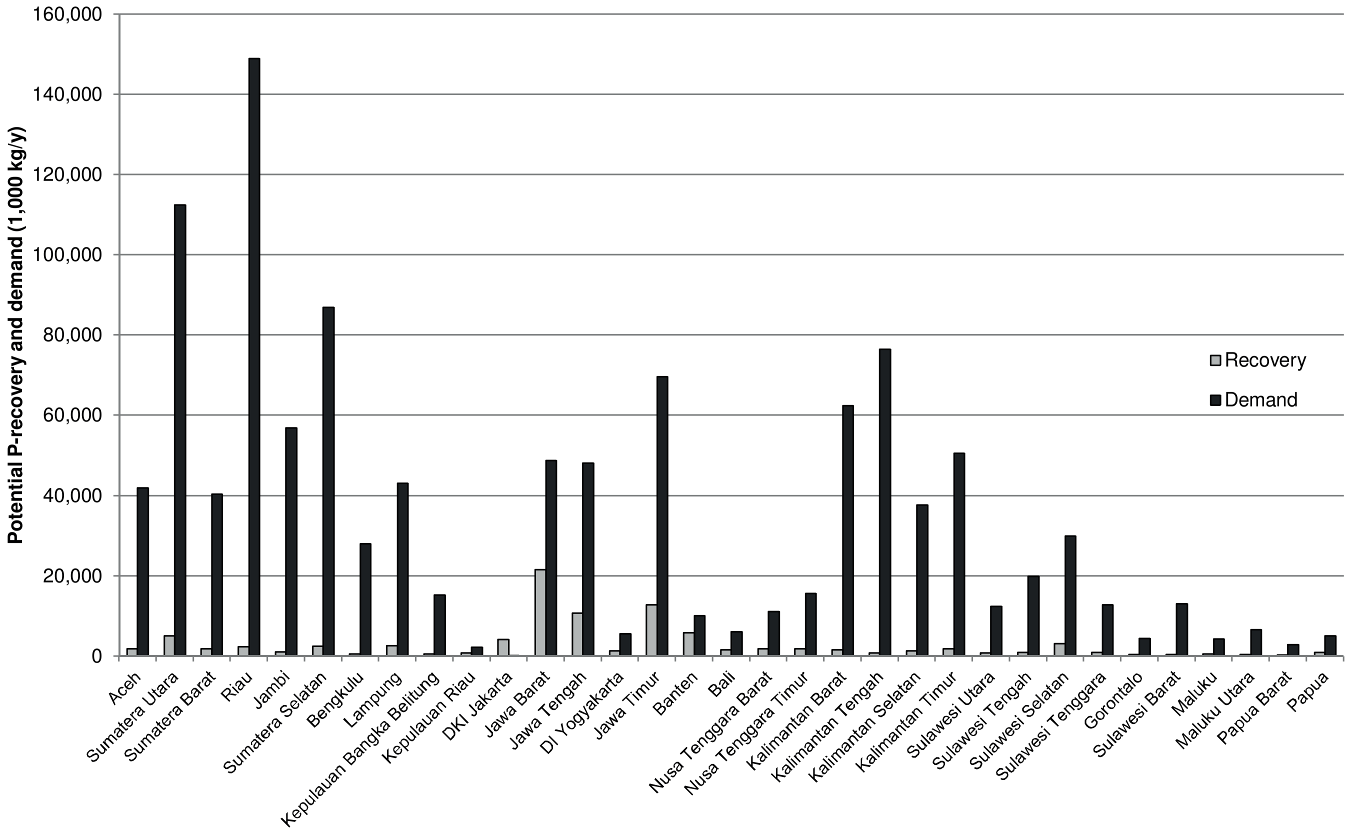

Figure 8.14 Province based (2035): P recoverable from on-site CBS and off-site WWTP (assuming AGS + 3R) and 100% 3R MSW scenario in relation to the actual demand

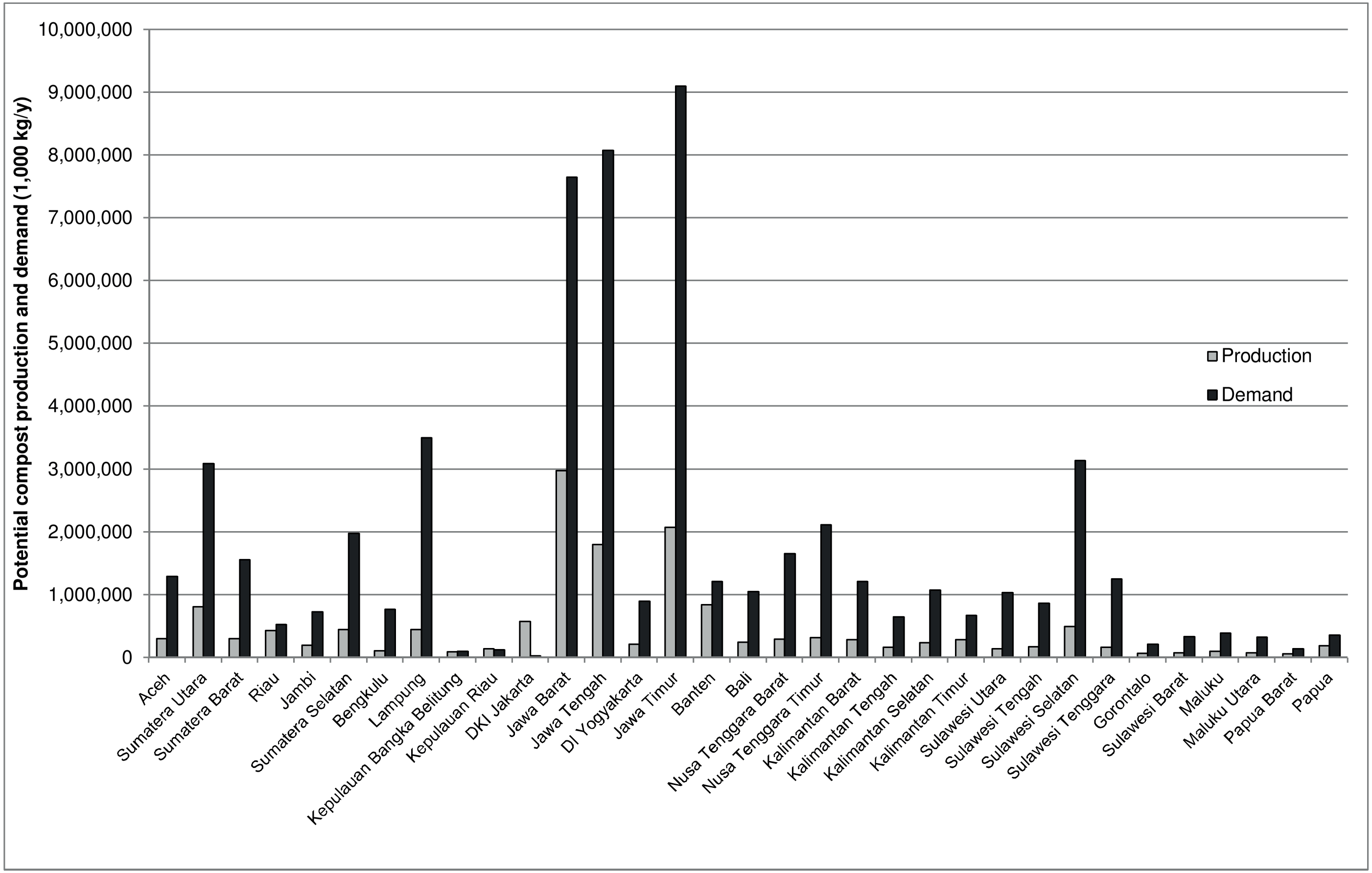

Figure 8.15 Province based (2035): Compost production from WWTP (AGS + 3R) and 100% 3R MSW scenario in relation to the demand

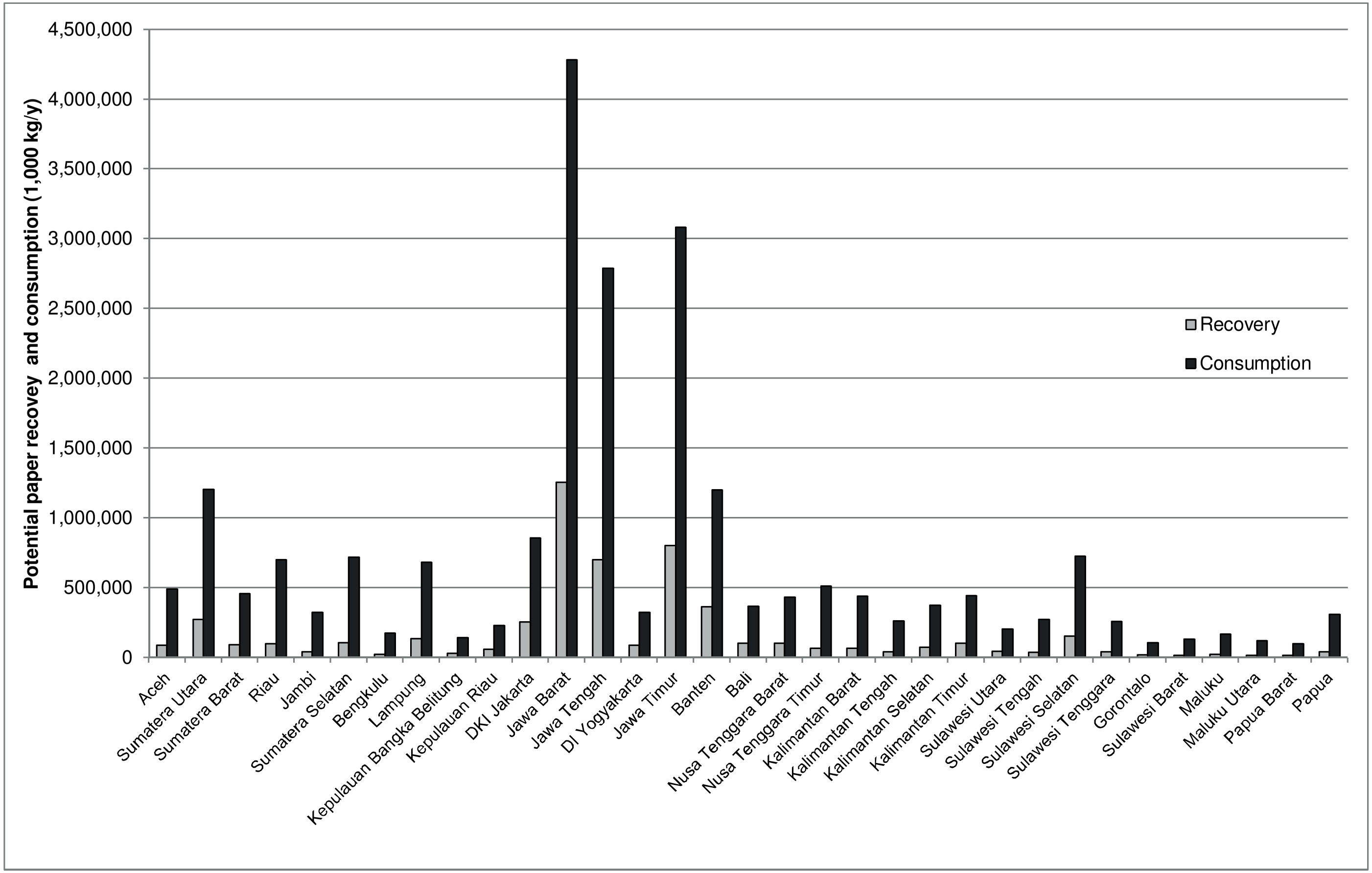

Figure 8.16 Province based (2035): Recoverable paper in relation to paper consumption of 100% 3R MSW scenario

The potential paper recovery (Figure 8.16) is highest in urban areas (Java) and lowest in rural areas (e.g. Papua). As explained in Chapter 5, the current implementation plan does not foresee plastic or paper recovery in low density (< 25 pp/ha) rural areas due to the remoteness of these areas. However, still nearly 25% of all consumed paper can be supplied from recovered paper. Currently, plastic and paper recovery processes are predominantly established in Java (see also Chapter 6). Insight in potential recovery and demand of plastics and paper can facilitate discussions between sanitation planners and industries to align activities for collection, recovery and trade flows of these products.

Figure 8.17 shows the potential cumulative supply of recovered resources from wastewater and solid waste activities compared to their potential demand for six clustered regions (Sumatra, Java, Kalimantan, Sulawesi, Bali & NTT & NTB, Papua & Maluku; see also Chapter 6).

Figure 8.17 provides an outlook on the extent of resource recovery from wastewater and solid waste and their contribution to material cycle closing for each identified region. It shows to what extent these regions could become independent of external resources and allows evaluation of the need to implement resource recovery technologies to develop a circular economy. Especially in Java, the potential to recover resources is considerable, and P and compost recovery potentials exceed 30%. Since both the long-term availability of worldwide phosphorus (Cordell et al., 2011) and to a lesser extent organic soil in Java (Minasny et al., 2011) are under pressure, insight in the spatial and temporal resource recovery potential from wastewater and solid waste may contribute to securing food systems. It may be an incentive for policy makers to apply a certain type of WWT or MSW technology to assure long-term availability of scarce (locally available) resources.

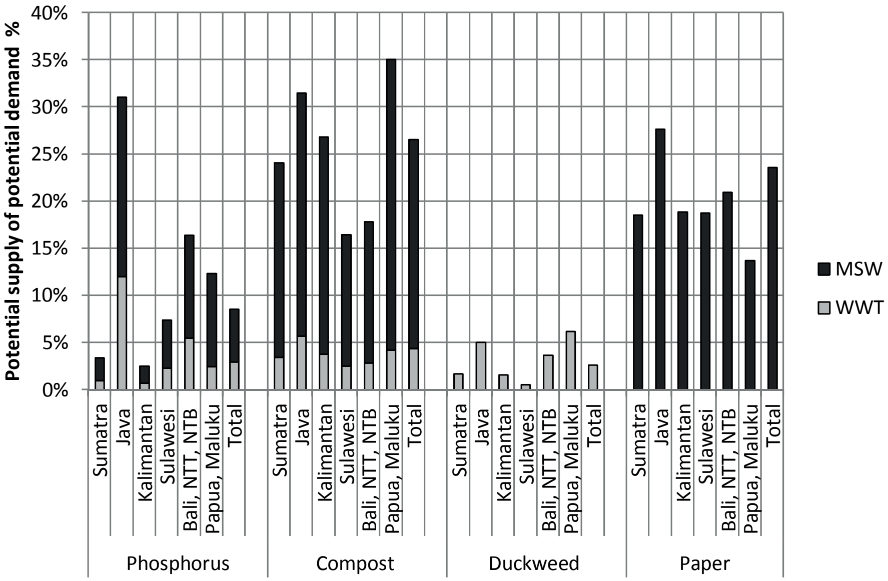

Figure 8.17 Potential supply compared to potential demand of phosphorus, compost, duckweed and paper) demand per region in 2035.The phosphorus and compost supply from wastewater is calculated assuming the use of the AGS + 3R off-site technology, whereas the potential duckweed supply it assumes the application of the UASB-DW-RBC

8.4.5 Can resource recovery accelerate access to sanitation?

It was hypothesized, that access to sanitation (wastewater and solid waste) facilities could be accelerated by considering resource recovery systems and technologies in the planning process. A demonstrated attractive BCR of bigger than 1 can feed into advocacy efforts to raise funding for sanitation development from governments and households. Once the private sector is convinced these players are ready to invest, the diverse funding sources and innovative capacity of the private sector can be unleashed (Hutton, 2013).

In Chapter 7, a BCR analysis for a specific Indonesian case (Upper Citarum River in West Java) was performed. This study showed that benefits through the sale of recovered resources from solid waste and wastewater represent a potential additional driver for improving water quality. These outweighed the additional costs for resource recovery facilities. It thus confirmed our hypothesis, namely that the BCR of sanitation interventions can be increased by applying resource recovery. In the study presented in Chapter 7, the benefits (health, access time, improved water sources and environment and increased land value) were not related to the resource recovery potential. Hence, the increased BCR was solely dependent on additional resource recovery benefits that exceeded the additional costs of resource recovery systems.

On a nationwide scale, these findings are the same, since addition of resource recovery technologies to CAS, AGS or MBR technologies could lower the total lifecycle costs (Figure 8.6 and Figure 8.7) of WWT while achieving the same effluent quality. At the same time, it was shown for WWT systems that these differences were only a few (1-5%) percentage points (Figure 8.6 and Figure 8.7). A much lower TLC (80% reduction) could be achieved by applying the low cost anaerobic filter technology. However, this would result in an increased discharge of pollutants and pathogens to the environment (Figure 8.10). The analysis in Chapter 7 (Upper Citarum River) showed that implementation of only low cost (anaerobic filter) technologies would not result in the target water quality. In addition, the added financial value of 3R in solid waste management was clearly demonstrated (Figure 8.8) and has been the reason for a lively, but informal solid resource recovery sector in Indonesia (Sasaki & Araki, 2013; Chaerul et al., 2013).
Moreover, it was demonstrated that the demand for recovered resources typically exceeds the supply (Figure 8.17), enabling selective marketing of resources, focusing on safe use (WHO, 2006). Heavily urbanized areas (e.g. Jakarta), showed an opposite trend in which the potential recovery of phosphorus, and compost from wastewater and solid waste exceeded the demand (Figure 8.14, Figure 8.15). These areas can therefore be regarded as urban harvesting areas (Agudelo-Vera et al., 2011) and may “feed” recovered resources to neighbouring provinces with a higher demand than supply (e.g. Banten, West Java).

8.5 Evaluation of the SaNaP framework

8.5.1 Added value of a new integrated approach

In this thesis we developed a new Sanitation Nationwide Planning (SaNaP) framework that can be used by policy makers for planning and evaluation of sanitation systems and considers local conditions (e.g. residential features) and demands (e.g. demand for resources).
The complexity of sanitation planning does not allow for the application of a single method or tool, but requires an integrated approach. This offers the possibility to have a nuanced view on interrelations compared to single cause-effect relations (Mirakyan & De Guio, 2013). The integrated evaluation of interventions provides significant benefits to projects in a dynamic context as was shown in the comparison of the impact of different systems on (1) costs (Figure 8.7) (2) pollution discharged (Figure 8.10) and (3) resource recovery potential (Figure 8.12) (Pollack, 2009). The presented framework enables evaluation of different criteria (e.g. costs, pollution loads, recovery potential) on different spatial (e.g. national, provincial and city) and temporal scales (2015-2035).
The framework was developed to make a planning and prepare a budget for wastewater and solid waste facilities in developing countries following governmental policy targets. The Indonesian situation, as an example for many developing countries, is characterized by inadequate sanitation regulatory frameworks and cross-sector policy coordination, rapid

urbanization, low community awareness on the importance of sanitation, limited land availability and local capacity to assure operation of facilities, and inadequate investments in sanitation systems (ADB, 2013; Kearton et al., 2013). Therefore, the framework acknowledges the need for a comprehensive situation analysis (Törnqvist et al., 2008) and identifies the main stakeholders and implementing institutions (BAPPENAS, 2007), applicable legislations (ADB, 2013), past sanitation development (WHO & UNICEF, 2014), and population development forecasts (BPS, 2013).

The involvement of stakeholders is an essential element in sanitation planning (UNEP, 2004; Ulrich et al., 2009; Van Buuren, 2010; Murray & Ray, 2010b) and enables a holistic view that otherwise may not be achievable (Thabrew et al., 2009). In the different stages of the development of the framework, a variety of stakeholders were involved to cover and address the issues debit to the backlog in sanitation development in Indonesia. These were (1) ministries (Planning, Public Works, Home affairs, Health and Agriculture), (2) representatives of international donors, such as the World Bank and WSP (Water and Sanitation Program of the World Bank), (3) NGO's active in the Indonesian sanitation development work, such as Bremen Overseas Research and Development Association (BORDA), HIVOS & SNV, (d) universities (Padjajaran University and ITB Bandung) and (4) organizations representing potential reuse stakeholders (Kemenperin, 2012; FAO, 2014b). In addition, SaNaP considers a variety of population groups (urban/rural, poor and non-poor) as important stakeholders, which is different to many other tools (Törnqvist et al., 2008; Mehta & Movik, 2010; Sijbesma, 2011),.

In order to increase the effectiveness and sustainability of sanitation interventions both "hardware" (e.g. sewer systems, collection vehicles, treatment systems) and "software" measures (e.g. campaigning, advocacy, institutional capacity building and technical assistance) are required (Waddington & Snilstveit, 2009). SaNaP uses the situation analysis and information from stakeholders to directly identify and quantify these "software" and "hardware" measures (Chapter 5). Sustainability is further assured by defining required implementation and operational budgets per responsible institution (Iyer et al., 2005; Mara et al., 2010; Winters et al., 2014). Besides, SaNaP quantitatively evaluates the sustainability of resource recovery, following a technical, financial, and resource flow (supply and demand) analysis.

The sustainability of SaNaP is further enhanced by a newly developed wastewater and solid waste systems selection framework (Figure 8.1) that uses key residential criteria (urban/rural features and population density) in a developing country's context. The use of residential criteria was based on their frequently readily and freely availability (BPS, 2014; DSM, 2014; NBSC (National Bureau of Statistics in China), 2014). The feasibility of certain technologies may vary in time and place from those pre-selected for Indonesia (Chapter 4). Therefore, in due course, more innovative technologies, e.g. those applying source separation and enhanced nutrient recovery (Larsen et al., 2009) or those that have a high degree of flexibility (Spiller et al., 2015), can be added to the framework to further improve its functionality and applicability (see also Section 8.5.4).

Sanitation infrastructure is often planned to last for a long time and are related to long-term policies (Ng et al., 2014). Also the ability to evaluate sanitation alternatives across spatial and temporal scales was identified as an important need in sanitation planning (Guest et al., 2009). Therefore SaNaP identifies both short (5 years) as well as mid- and long-term interventions (up to 20 years) in its planning. Spatial evaluation is enhanced by the use of data collected at the smallest administrative unit (*desa*) that can be aggregated to higher level of governments. For Indonesia the raw data of nearly 80,000 *desa* was collected to develop SaNaP. The spatial evaluation supports the identification or formulation of location specific interventions that considers the level of, for example, poverty and urbanization as well as resource demand (see also Figure 8.10-Figure 8.17 and Chapter 5). SaNaP enhanced the spatial evaluation by visualization of planned infrastructures in a Geographic Information System (GIS), as was shown in Chapter 5, which is also found a novelty in sanitation planning.

8.5.2 Applicability of the framework in the field

SaNaP has been developed to guide policy makers in the preparation of nationwide budgets and to evaluate how certain choices (e.g. targets, priorities, system selection) affect the required budgets and implementation of facilities at different levels. The advantage of a residential area-based selection system is that most of the required input data is readily available through on-line databases (BPS, 2010, 2013, 2014) and does not require in-depth and resource or time demanding local surveys. Because residential data is collected on a local scale, the determined budgets and number of systems can be easily translated to local implementation (Chapter 5).

A second advantage of the method that enhances application is that the impact of system or technology choices can be evaluated not only on financial parameters (CAPEX, OPEX, TLC), but also on environmental parameters (pollutants and pathogens discharged), spatial parameters (land requirements) and resource (recovery) parameters (e.g. energy requirement, sludge production, duckweed production). This enables site specific selection of technologies. For example, if stringent effluent standards prevail (e.g. in high densely populated areas), the interventions can be tailored to select technologies that meet those requirements (e.g. CAS N&P, UASB-DW-RBC, MBR, AGS), while immediately quantifying the impact (costs, space and energy requirement, resulting pollution loads discharged) of these interventions. Alternatively, if an area features major aquaculture activities, SaNaP can evaluate how duckweed production from wastewater treatment can contribute to the required feed based on the number of people that need to be connected to the system.

In Chapter 3, it was described how the current sanitation policies favoured the application of DEWATS (community based sanitation, CBS) systems. SaNaP can be used to evaluate the impact of such policy choices, in terms of number of systems and land required, impact on pollutants discharged etc. For example, if all people that were planned to be connected to off-site systems (about 13% of the population in 2019 see Chapter 5) would connect to CBS systems, approximately 80,000 installations until 2019 are required. The described difficulties in operation and maintenance as well as space requirement in urban areas of these CBS (Eales et al., 2013) shows that the described policy and corresponding required implementation is not feasible. In addition, the use of CBS increases the P discharge in urbanized area by over 50% compared to nutrient removing systems, which was shown in Figure 8.10 B (Jakarta 2035 example; knowing that CBS and anaerobic filters show similar performances, as determined in Chapter 4).

Finally, the applicability of SaNaP in a developing context is shown in the formulation of the Indonesian "National Medium Term Development Plan (2015-2019)", in which SaNaP was used to determine the required budget to meet the sanitation targets. To be eligible for national funding, Indonesian cities must prepare a City Sanitation Strategy (CSS). In this 5-year plan, locally required budgets and institutional strengthening and advocacy and campaigning activities are formulated (Kearton et al., 2013; Parkinson et al., 2014). The CSS is based on a similar residential-area dependent system selection and budget formulation of WWT and MSW systems and over 400 (on a total of 507) cities started their CSS by 2013 (USDP, 2014). This shows the applicability of SaNaP for both local and national governments.

8.5.3 Applicability of SaNaP in other countries

The selection approach for a WWT or MSW system based on residential features can be used for planning purposes not only for Indonesia, but for developing countries in general. The typical performance and per capita resource (e.g. energy, sludge, space) consumption and production data are considered representative for similar developing countries (e.g. South East Asia, South America) or could be updated with local information. Following a cost update on investments and operation unit costs parameters, the presented approach can be applied for WWT and MSW system selection in other developing countries facing similar challenges as Indonesia.

Further, population and residential area development are readily available through various data bases (DSM, 2014; NBSC, 2014), and UN (United Nations)-reports (UNpopulation, 2012; WHO & UNICEF, 2014). The division of responsibilities and budgets per type of intervention applying the matrix as presented in Chapter 5 can be replicated in any country. Table 8.1 shows such typical matrix and how it can be developed by determining the percentages for source funding (now indicated with an X). The activities (e.g. studies, designs), responsible institutions (e.g. ministries, private parties) and cost type can be adjusted depending on the local practice in that country. For example, if a considerable part of funding is obtained from foreign NGO's, this should be included as well.

Table 8.1 Required Division (%) of budget [a] per (i) level of funding, (ii) department and (iii) activity. This Table has been slightly modified from the examples presented in Chapter 5.

Sub-sector	Wastewater or Solid waste systems											
System	System 1				System 2				System 3			
Activity	%	source			%	source			%	source		
		i	ii	iii		i	ii	iii		i	ii	iii
Studies and design												
Master plan, Feasibility analysis									X	N	PW	S
Environmental impact assessment									X	N	PW	S
Guidelines and detailing									X	Lo	PW	S
Campaign, Advocacy, institutional strengthening												
General	X	N	CAI	S	X	N	PW	S	X	N	CAI	S
Local	X	Lo	CAI	S	X	Lo	CAI	S	X	Lo	CAI	S
Land					11	U		La	X	Lo		La
Construction												
House connection	X	U		H	X	U		H	X	Lo	PW	H
Collection	X	N	PW	H	X	N	PW	H	X	N	PW	H
Treatment	X	U		H	X	N	PW	H	X	N	PW	H
All	100				100				100			

[a] Source codes i, ii and iii refer to:

iv. Level of funding: national (**N**) or local (**Lo**) government or users/private (**U**);

v. Ministry: Public Works (**PW**), Ministry of Health/Home Affairs (**CAI**) for campaign, advocacy and institutional strengthen;

vi. Type of activity: hardware (**H**), software (**S**) or land acquisition (**La**)

The determination of resource demand was based on publicly available data from the statistical bureau of Indonesia and FAO, supplemented with country specific fertilizer practises and interviews with local stakeholders (ministries, associations and universities). This makes the methodology for the spatial resource demand for crop nutrients, proteins and recoverable plastics and papers applicable for other countries as well.

8.5.4 Options for extending SaNaP

In the illustration of this framework we focused on technologies that fit the current Indonesian context. For that reason, new sanitation systems that aim to recover resources following source separation were excluded (Otterpohl et al., 1997; Larsen et al., 2009; Zeeman & Kujawa-Roeleveld, 2011; Tervahauta et al., 2013). In Chapter 2, the application of new sanitation in a new Chinese residential development was studied and showed that such systems can be very attractive in terms of finances and waste valorization. A first extension option for future scenarios (e.g. Indonesia after 2035), or for countries in a different development stage (e.g. the Netherlands or Singapore), is the inclusion of other or more technologies. All technologies can be accommodated in the framework as long as their feasibility is linked to residential features and the investment and operational costs are known (see Figure 8.1). Further, Spiller et al. (2015) propose implementation of technologies and infrastructure which are flexible, adaptive and robust in order to ensure the sustainability of these systems under dynamic conditions. These may comprise modular or prefabricated systems that can cope with changing capacities or the addition of a post treatment step for nutrient removal or disinfection to meet future effluent requirements. In the current framework for Indonesia this was already considered for urban slums. These slums would initially apply temporary community sanitation centers (Ulrich et al., 2009), since slums often lack water facilities, consist of temporary or non-legal houses, and their residents are unable or unwilling to pay for off-site systems (Sijbesma, 2011). The temporary solutions can be replaced by or adjusted to a more structural solution during planned renovation or rehabilitation (Bappenas, 2014; USDP, 2015).

Second, system selection criteria can be extended. Currently, SaNaP uses residential features to select systems, since these reflect the impact that available systems may have on public health and the environment (Chapter 4 and 5). Besides population density and urban features, applicability of systems is determined by other factors as well, such as ground water levels, soil conditions and availability of a (piped) water supply (Loetscher & Keller, 2002). In addition, the practical suitability of systems can be included in the final system selection. For example, in more remote areas skilled labour, spare parts or required energy and chemicals may be not available and systems depending on them are not appropriate (Senzia et al., 2003). Moreover, successful operation of facilities depends on the institutional and management capacity of the responsible actors (see Chapter 5 and Table 8.1). Therefore, additional criteria that are available on a nationwide scale can be incorporated to improve the applicability of the framework. Possible criteria on a nationwide scale that reflect additional system selection criteria are the availability of electricity, piped water supply or general level of education (Ministry of Health, 2013). Besides the use of data collected though surveys, the use of “big data” can be considered as an indication for sanitation needs. Big data is data that is typically created digitally (e.g. social media), passively produced as a by-product of our daily lives, automatically collected, geographically or temporarily trackable and continuously analyzed (UN Global Pulse, 2012). The use of “big data” is considered a genuine opportunity to bring powerful new tools to the fight against poverty, hunger and diseases, since it allows to turn imperfect, complex, often unstructured data into actionable information (UN Global Pulse, 2012). Big data has been successfully applied in the development of flood management strategies (Jongman et al., 2015). Big data has also been used to prepare a flooding profile for Jakarta, using frequency and location of tweets including words, such as “flood”, and “water damage” (social data mining)

(Wagemaker & Loenen, 2014). Similarly, social data could be mined to prepare sanitation profiles in an area using sanitation related criteria, such as e.g. garbage, diarrhoea, or smell.

Third, the current framework predominantly focused on domestic wastewater and solid waste streams, whereas SaNaP can be extended using other sources of pollution. In Chapter 7 (Upper Citarum River) specific attention was already paid to industrial pollution and this showed the need to develop the domestic and industrial environmental WWT sector in parallel to meet the target water quality. Therefore, for specific cases integration with industrial pollution actors (following the example of Chapter 7) can be considered to come to an effective approach to improve public health and the environment.

Fourth, the objective of sanitation is to prevent pollution and improve public health. However, it is necessary to assess sustainability of interventions in order to ensure that pollution is being removed and not displaced. Similar to the analysis performed for Changzhou (Chapter 2), the inclusion of environmental emissions other than water pollution (COD, N & P) may enhance sanitation planning. Examples of such emissions are odor or greenhouse gasses (Rodriguez-Garcia et al., 2011). Technologies that may be low cost (e.g. anaerobic ponds, septic tanks) emit considerable greenhouse gasses and odour, whereas the current SaNaP does not consider these effects yet. Wastewater and solid waste system emission values of greenhouse gasses or odorous compounds, such as sulphide, are available (Aye & Widjaya, 2006; Zitomer et al., 2008; Kerstens, van der Steen, et al., 2009; Larsen, 2011) and can be incorporated in SaNaP.

Fifth, in the determination of the BCR (Chapter 7) not all economic impacts were quantified, such as consumption of fish imbibing toxic wastes or otherwise infected (Lasut et al., 2008), reduced land subsidence resulting from reused industrial effluent, improved recreational values (Day & Mourato, 1998; Alam, 2008), reduced industrially discharged toxins and heavy metals or anticipated long-term effects of reduced eutrophication and less impacted ecosystem functioning (Suwarno et al., 2013). Monetizing these impacts may further increase the benefit of sanitation interventions and enhance advocating sanitation interventions.

Sixth, SaNaP can be extended by elaborating on specific market demands for recoverable resources. In our study we compared the potential supply with potential demand, without an in depth analysis of local availability of alternative resources and the recovery costs compared to prices of competitive resources. Moreover, to further safeguard the quality and safety of produced or recovered products (Snyman & Vorster, 2011; Raschid-Sally, 2013), specific attention can be paid to required source (e.g. need for source separation) and level of hygienization of sanitation by-products and the perception of envisaged users (Koné et al., 2007). Thus far, studies show ambiguous demand for sanitation by-products by farmers (Starkl et al., 2010; SNV, 2013) despite their potential financial benefits. Further, the quality of recovered raw materials (e.g. plastics) largely determine their potential to become a replacement of virgin plastics (Lazarevic et al., 2010). SaNaP provides an initial assessment of the possibility to move towards a circular economy, but to foster long-term sustainability the quality and marketing of

recovered products need to be matched with specific local demands and customs (Cordell et al., 2011; Saveyn & Eder, 2014; Diener et al., 2014). In the provided analysis (e.g. Figure 8.17) local (national) resource demand and supply were compared, whereas international or regional trade patterns were only partly included (e.g. for waste paper in Chapter 6). Established international trades of recoverable resources such as for phosphorus, paper and plastics can provide alternative and potentially more attractive financial markets (Van Beukering, 2001).
Next, SaNaP can be extended by including more types of recoverable resources. The selection of resources (Chapter 6) was based on formulated criteria, such as the recovery potential from wastewater and solid waste in the Indonesian setting, fraction of solid waste and transportability. However, there are a number of other resources or products that can be recovered from wastewater and solid waste for which markets exist. For example, nitrogen and potassium (present in wastewater) are essential nutrients for crop development (Janssen et al., 1990). Current nitrogen fertilizer production is an energy intensive process (Tervahauta et al., 2013). However, new technologies (e.g. microbial fuel cell) allow for the recovery of ammonium-nitrogen and simultaneously produce energy from concentrated (e.g. urine) waste streams (Kuntke et al., 2012). Also potassium can be well recovered from source separated domestic wastewater streams (Zeeman & Kujawa-Roeleveld, 2011). Since (pre)-treated effluent may contain high levels of nutrients (see Chapter 4, Table 4.4), its direct use for irrigation purposes is increasingly attracting the attention of policy makers, officials and researchers (Huibers & Van Lier, 2005). The benefits of using (pre)-treated effluent should be offset against potential barriers, such as health issues (WHO, 2006; Yaya-Beas et al., 2015), unbalanced nutrient content of effluent in relation to the crop demand (Janssen et al., 2005), legislation (MoPW, 2001), and the perception of farmers and consumers (Starkl et al., 2010). In Chapter 2, we proposed the reuse of effluent for greening only after enhanced (membrane) treatment to comply with the applicable legislation. In Indonesia, reuse of (pre) treated wastewater is restricted by nutrient content and pathogen levels (MoPW, 2001), as was shown in Chapter 4, Table 4.4. Moreover, in the current analysis (Chapter 6) duckweed production as a feedstock for aquaculture was assumed following successful experience in the field (Journey et al., 1993; Islam et al., 2004). At the same time, duckweed is also considered as a replacement for animal-derived proteins to enter the European market (van der Spiegel et al., 2013). In addition, duckweed may be used as a source for bio-fuel production and is regarded a promising alternative for bioenergy production (Cheng & Stomp, 2009; Verma & Suthar, 2015). Likewise, algae production from wastewater has been widely applied (Chernicharo, 2006; Laxton, 2010) and may be used for protein and biofuel production as well (Adenle et al., 2013). Following the determination of specific energy or protein production from harvested duckweed and/or algae per person, and the corresponding monetized value, implementation in a sanitation planning can be realized (see also Figure 8.5). Glass (2%) and metals (4%) are present in much smaller fraction in domestic solid waste than organic matter (59%), plastic (14%) and paper (12%) (Aprilia et al., 2011) and were therefore excluded from

analysis presented in Chapter 6. However, the recovery of these materials might still be attractive, especially considering construction waste streams (Tam & Tam, 2006). Thus, similar to the selected resources (Chapter 6) that were used to illustrate the functioning of the framework, alternative resources can be included to extent SaNaP and to determine the nationwide recovery potential in relation to its demand.

Finally, SaNap may be enhanced by extension of visualized geographical related output. Spatial scales are considered in SaNaP and visualization of interventions (location of systems) in Geographical Information Systems (GIS) was illustrated (Chapter 5). A regional based output of recoverable P and compost was further visualized in a map in Chapter 6. Resource recovery potentials and consumption parameters (energy, phosphorus, compost, sludge, land, chemicals, water) were determined on the *desa* (smallest administrative unit) level, which would allow to visualize the resource harvest potential in GIS within a city district (*kecamatan*), at city level or on a provincial level. Thus, the resource recovery as a percentage of the demand values, e.g. as presented in bar diagram for clustered regions in Figure 8.17, can be mapped in GIS on a provincial level as well. The visual presentation of these resource supply and demand parameters can further support regional priority setting, selection and use of recovery technologies (Quaye-Ballard & An, 2010; Coutinho-Rodrigues et al., 2011).

8.6 The way forward

The absence of sanitation facilities negatively impacts millions of lives worldwide, especially in developing countries. Planning, implementation and operation of sanitation infrastructures has shown to be a major challenge. The anticipated population growth and urbanization will only further complicate the sanitation challenge, impacting vulnerable population groups most. The developed Sanitation Nationwide Planning Framework provides guidance in the development of sanitation infrastructures and aims to accelerate the number of people that have access to sanitation in developing countries. It further showed how resource recovery from wastewater and solid waste may be a driver in accelerating access to sanitation.

The here presented framework was a cornerstone in the formulation of the Indonesian “National Medium Term Development Plan (2015-2019)”. The application of this framework in other rapidly developing countries may benefit the quality of life of millions of people.

8.7 References

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Summary

Worldwide 2.5 billion people lack access to sanitation. This impacts public health, environment, welfare, and moreover results in a loss of resources. Conventional sanitation systems consume energy, chemicals, land and produce sludge that requires disposal, whereas a range of opportunities exists that enables valorization of resources from our "waste", such as energy, phosphorus, compost, plastic and paper. Resource recovery may become a driver for economic growth and respond to profound changes of the world's population impacting food security and availability of finite natural resources.

The backlog in sanitation development can partly be attributed to the absence of a functional sanitation planning framework that allows for integration of cross-sectoral elements, such as health, technical, environmental, financial, institutional, demand for sanitation by-products, and welfare aspects. To evaluate a set of alternative sanitation systems, policy makers require a framework for resolving trade-offs, costs and benefits across spatial and temporal scales, and sustainability dimensions (social, environmental and economic).

In **Chapter 2** we demonstrated that the integration and material cycle closing of water, waste and energy in a Chinese residential area development (Qinglong district in Changzhou) will become beneficial to the establishment of the envisaged green city. Four different scenarios focusing on water, nutrient and energy recovery were compared with the baseline wastewater management practice. Besides environmental benefits, the economic benefits of the resource recovery oriented sanitation concepts were shown. The financial break-even point with the baseline scenario was already after 5 years, provided that recovered resources can be sold for a marketable price. The presented concepts were considered to be applicable for a wide range of new urban developments in China and similar rapidly developing densely populated regions worldwide.

Despite the potential benefits of resource recovery oriented sanitation concepts, developing countries often do not consider alternative sanitation systems or integrate identified cross-sectoral elements to select a sanitation system. Rather, policies promote the introduction of a single type of system only. In Indonesia, for instance, decentralized (communal) wastewater treatment systems (DEWATS) are promoted as the core of the sanitation improvement. Under the Indonesian "Accelerated Sanitation Development for Human Settlements Program" thousands of new DEWATS are planned for construction in the coming five years. In **Chapter 3** we therefore evaluated the technical and financial-economic aspects and users' involvement of three different DEWATS: (1) Settler + Anaerobic Baffled Reactor (ABR) + Anaerobic Filter (AF), (2) Digester + Settler+ ABR + AF, and (3) Settler, equalization, activated sludge, clarifier and filtration. The evaluation showed that all three systems complied with the current regulations. Further, a clear reduction in specific investment costs

per household was found with an increasing number of households per system. This shows the potential of scaling up community based systems (typically 100 households per system) to medium centralized off-site systems (typically 500-5,000 households per system). Only regular operational costs (e.g. wage of the operator) were recovered from fees collected by the community, whereas costs for desludging, major repairs and capital and replacement costs were not. Surveys with users showed different levels of involvement of local men and women in the planning stages of the project. The study recommended that application of DEWATS should be evaluated in the context of a (city wide) sanitation strategy.

In spite of existing sanitation system selection criteria and the demonstrated link between residential features and occurrence of health and environmental issues in the absence of sanitation, an integrated sanitation systems analysis for different residential conditions is lacking. To develop a sanitation planning framework, first a technical and financial feasibility analysis of wastewater and solid waste systems for application in Indonesia was prepared**.** **Chapter 4** describes the selection of on-site, community-based and ten off-site wastewater systems as well as conventional, centralized and decentralized 3R (Reduce Reuse Recycle) solid waste systems. COD, BOD, nitrogen, phosphorus and pathogen removal efficiencies, energy requirements, sludge production, land use and resource recovery potential (phosphorus, energy, duckweed, compost, water) of wastewater treatment systems were determined. Solid waste systems were analyzed according to land requirement, compost and energy production and recovery of plastic and paper. In the financial analysis, investments, operational costs and benefits and Total Lifecycle Costs of all investigated options were compared. Technical performance and TLC were used to guide system selection for implementation in different residential settings. The effect of price variations of recoverable resources and land prices on total lifecycle costs was determined in an analysis. A 10-fold increase in land prices for land intensive wastewater systems resulted in a 5 times higher TLC, whereas a 4-fold increase of the recovered resource market price resulted in maximum 1.3 times lower TLC. For solid waste, these impacts were reversed – land price and resource selling price variations resulted in a maximum difference in TLC of 1.8 and 4 respectively. Technical and financial performance analysis can therefore support decision makers in system selection and anticipate the impact of price variations on long-term operation.

To translate government policy in sanitation implementation strategy, the outcomes of the performed feasibility analysis was incorporated in a sanitation planning framework. Available sanitation planning frameworks were not found applicable, since these did not include (1) all population groups, (2) both wastewater and solid waste treatment and resource recovery systems, (3) readily available selection criteria, (4) integration with land use planning activities, and (5) identification and budget allocation of implementing institutions. Therefore, in **Chapter 5** a comprehensive framework was developed that directly links a government policy to a nationwide long-term planning and budgeting for wastewater and solid waste interventions. The framework requires input from different stakeholders, such as government planners and experts to formulate starting points and targets. Based on a limited number of indicators to enable the sanitation system selection (population density, urban functions), three outputs are generated. The first output is a selection and visualization of the spatial distribution of wastewater and solid waste systems. The second output generates the total number of people served, budget requirements and distribution of systems. Thirdly, the required budget is allocated to the responsible institution to assure effective implementation. The determined budgets are specified by their beneficiaries, distinguishing urban, rural, poor and non-poor households. The framework was applied for Indonesia and outputs were adopted in the National Development Plan. A more than fivefold increase of the national contribution as compared to the current budget allocation is needed for Indonesia. The budget for campaigning, advocacy and institutional strengthening to enable implementation was determined to be 10% of the total budget.

The initial objective of the developed sanitation planning framework is to accelerate access to sanitation. Therefore, it primarily focuses on the beneficiaries (or “front-end” users) of sanitation facilities. However, to foster long-term operational and financial sustainability, also the needs of potential “back-end” user of sanitation products should be considered. Back-end users comprise among others agriculture, horticulture, aquaculture and plastic and paper processing industries. Despite the availability of methods to analyze material flows and demand forecast, a comprehensive framework that includes recoverable resources from both wastewater and solid waste and that allows for a nationwide temporal and spatial demand forecast is lacking. Therefore, in **Chapter 6** the future potential demand of recoverable resources based on past consumption trends and future forecast for a selected number of recoverable resources is described. Phosphorus and compost demand analysis was based on (1) fertilizer requirements of 68 staple foods, horticulture and plantation crops and (2) anticipated increase in production area of these crops. Duckweed demand as a protein-rich fish feed was analyzed based on the forecasted demand from *tilapia* and *carp* aquaculture. The potentially recoverable (waste) plastic and paper to substitute conventional manufactured products were based on extrapolation of past trends in plastic and paper production in Indonesia. The potential contribution of recoverable products to the forecasted demand for 2035 was assessed for phosphorus (15%), compost (35%), duckweed (7%), plastic (66%) and paper (18%). A geographical discrepancy between potential recovery and demand location for phosphorus and compost was determined. Therefore, the locations of potential markets should be considered in the planning and selection of wastewater and solid waste facilities.

Following the developed sanitation system selection criteria, planning framework, and resource demand analysis, **Chapter 7** describes a methodology to support a policy maker in formulating a cost- and environmentally effective sanitation strategy. This required an analysis of (i) sources of pollution, (ii) mitigating measures and resource recovery potentials and their effect on health and water quality, and (iii) benefits and costs of interventions. The impact of different sanitation interventions on (1) water quality improvement, (2) resource recovery potential, and (3) monetized benefits to costs ratio were quantified. The Benefit Cost Ratio (BCR) compared monetized benefits (health, access time, improved water sources & environment, land values and sale of recovered resources) to required costs of interventions (CAPEX and OPEX). The integration of technical, hydrological, agronomical and socio-economic elements to derive these three tangible outputs in a joint approach is a novelty. The applicability and added value of this approach was demonstrated using the heavily polluted Indonesian Upper Citarum River in the metropolitan Bandung – Jakarta region. Domestic interventions, applying simple (anaerobic filter) technologies were economically most attractive with a benefit cost ratio (BCR) of 3.2, but could not reach target water quality. To approach the desired water quality, both advanced domestic (nutrient removal systems) and industrial wastewater treatment interventions were required, leading to a BCR of 2. Benefits from selling recovered resources from solid waste and wastewater represent here an additional driver for improving water quality and outweigh the additional costs for resource recovery facilities. It was thus shown that water quality interventions justify their costs and are socially and economically beneficial.

In the discussion **Chapter 8**, the Sanitation National Planning framework (SaNaP) is presented. The potential of the SaNaP to evaluate system costs, pollution loads, production and consumption parameters and potential resource demand and supply for a set of alternative sanitation systems is illustrated using Indonesia as an example. The introduction of resource recovery concepts in the Indonesian sanitation sector development can contribute considerably to a circular economy. For Java that accounts for nearly 60% of the Indonesian population, one third of the compost and phosphorus demand can be satisfied through recovered resources. Resource recovery is shown to be a potential driver to accelerate sanitation development. Several possibilities are identified to enhance the functioning of SaNaP, such as extension of the number of (1) included technologies, (2) system selection criteria, (3) environmental indicators, (4) monetized benefits, (5) specific market demands, (6) recoverable resources, and (7) visualized geographical related output. The here presented framework was developed for the Indonesian government, but the application of this framework may benefit the quality of life of millions of people in other rapidly developing countries.

List of abbreviations

3R: Reduce Reuse Recycling of solid waste, but also applied to show that recovery of resources is applied for wastewater treatment technologies
ABR + AF: Anaerobic Baffled Reactor + Anaerobic Filter
ADB: Asian Development Bank
AGS: Aerobic Granular Sludge
BOD: Biological Oxygen Demand
BPS: *Buro Pusat Statistik* (Central Statistical Bureau of Indonesia)
CAPEX: Capital Expenditures
CAS: Conventional Activated Sludge
CBS: Community based Sanitation
CLTS: Community Lead Total Sanitation
COD: Chemical Oxygen Demand
DKI: *Daerah Khusus Ibukota Jakarta* ("Special Capital City District of Jakarta"),
DSM :Department of Statistics in Malaysia
DW: Duckweed Pond
GIS: Geographic Information Systems
IPLT: *Instalasi Pengolahan Limbah Tinja*; Sludge processing facility
JSM: Java Spatial Model
MBR: Membrane Bioreactor
MDG: Millennium Development Goals
MFA: Material Flow Analysis
MoE: Ministry of Environment of Indonesia
MoPW: Ministry of Public Works (of Indonesia)
MSW: Municipal Solid Waste
N: Nitrogen
NBSC: National Bureau of Statistics in China
NPV: Net Present Value
O&M: Operation and Maintenance
ODI: Overseas Development Institute
OPEX: Operational Expenditures
OSWF: Organic Solid Waste Fraction
P: Phosphorus
RBC: Rotating BioContactor
Rp: Rupiah (currency applied in Indonesia)
SANIMAS: *Sanitasi oleh Masyarakat* (Community Based Sanitation)
SDG: Sustainable Development Goals
TLC: Total Lifecycle Costs

TN: Total Nitrogen

TP: Total Phosphorus

TTPS: *Tim Teknis Pembangunan Sanitasi;* technical team for sanitation development

UASB: Upflow Anaerobic Sludge Bed

UN: United Nations

USDP: Urban Sanitation Development Program

WHO: World Health Organization

WSP: Water and Sanitation Program of the World Bank

WWT(P): Wastewater Treatment (Plant)

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Kerstens, S.M., Hutton, G., Firmansyah, I., Leusbrock, I. & Zeeman G. An integrated approach to evaluate benefits and costs of wastewater and solid waste management to improve the living environment: the Citarum River in West Java, Indonesia

Others

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About the author

Sjoerd Kerstens was born in 1977 in Heerlen, The Netherlands. After having obtained the “Atheneum” diploma from the Sint-Jans College in Hoensbroek in 1996, he went to Wageningen to study Environmental Science. During his study he developed an interest for developing countries. From June 1999 to April 2000 he entered an internship to develop an Eco-tourism campsite in Kwazulu Natal (South Africa). From February 2001 to December 2001 he studied at the WERSC (Water and Environment Research and Study Center) at Jordan University to work on his Master thesis “Anaerobic treatment of strong sewage; One-stage and two-stage systems”. Following a second Master thesis at the laboratory of microbiology, he fulfilled his internship at the UNESCO-IHE institute in Delft where he studied the effect of a duckweed cover on sulphide volatilisation from waste stabilisation ponds. After his graduation (*cum laude*) in 2002 he worked as a researcher for LeAF (Lettinga Associates Foundation) until April 2003 after which he followed his wife Jessica to Beijing, China. From 2003 until 2007 he worked there as a process engineer for DHV on industrial wastewater projects and, from 2005-2007, as a water and sanitation expert on the FTEI project (Feasible Technology for Environmental Infrastructure in Western Chinese small towns). From 2007 until 2010 he worked at the industrial water treatment department at DHV in Amersfoort (The Netherlands). During that period he also became an authorized Cradle to Cradle consultant. From 2010 - 2014 he worked as water & sanitation specialist in Indonesia (Jakarta) on industrial water projects and, from 2011 onwards, on the Urban Sanitation Development Program (USDP) project. During that period he helped developing the sanitation decision support systems to guide over 400 cities in preparation of City Sanitation Strategies and prepared the National Sanitation Assessment study for the Ministry of Planning. In 2012 he started his PhD work (as an external candidate), in which he combined his working experiences with developments in the research fields of sanitation and resource recovery. He returned to the Netherlands in 2014, where he continued his work for Royal HaskoningDHV at the Process and Innovation department.

Sjoerd is an active football player with a rich national and international career, among which the finest of teams, such as RKBSV, GVC-Wageningen, KwaMvutshane FC (South Africa), Jordan University indoor soccer team, Beijing Drifters (China), FC Bugils (Jakarta, Indonesia) and the Cobu Boys (Amersfoort, the Netherlands). Thus far, no new teams have shown interest to sign him up... He has a long lasting passion for Roda JC and U2. He is very happily married to Jessica and they have three sons: Jelle (10 yrs), Arne (9 yrs) and Tijn (6 yrs).

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