Desalination and Water Treatment
www.deswater.com

doi: 10.5004/dwt.2017.21247

91 (2017) 12–22
October

The evaluation of technologies for small, new design wastewater treatment systems

G. McNamara[a,*], L. Fitzsimons[a], E. Doherty[b], E. Clifford[b], Y. Delaure[a]

[a]*School of Mechanical Engineering, Dublin City University, Glasnevin, Dublin 9, Ireland, emails: greg.mcnamara5@mail.dcu.ie (G. McNamara), lorna.fitzsimons@dcu.ie (L. Fitzsimons), yan.delaure@dcu.ie (Y. Delaure)*
[b]*College of Civil Engineering and Informatics, Ryan Institute, National University of Ireland, Galway, University Road, Galway, Ireland, emails, edelle.doherty@gmail.com (E. Doherty), eoghan.clifford@nuigalway.ie (E. Clifford)*

Received 24 December 2016; Accepted 21 July 2017

ABSTRACT

Life cycle costing of small wastewater treatment systems can often be generic and lack a degree of detail that could affect the choice of system. Critical factors such as variations in loading, location and discharge limits are sometimes not given the required weight of importance, and as a result the most suitable, most economical system may not always be implemented. A decision support tool for small, new design wastewater treatment plants has been developed that accounts for variations in several parameters such as scale, discharge limits and sludge disposal. Capital and operational costs have been combined to produce life cycle models for six treatment systems. Each system was assessed in a number of scenarios with variations in scale, discharge limits and sludge disposal route. The results show that in most scenarios, constructed wetlands represent the most economical option where surface area is not restricted. For each system, the percentage contribution of labour to the total operational cost increases as agglomeration size is reduced.

Keywords: Wastewater treatment; Capital and operational expenditure; Life cycle costs; Scenario-specific conditions; Discharge limits

1. Introduction

Small communities and rural agglomerations face multiple challenges in fulfilling their wastewater treatment requirements. In many cases, for both developing and developed nations, the main issue is limited capital resources. Public utilities' capital investment allocation is often prioritised for large agglomerations where the potential risk of environmental consequence is higher. There are a number of economies of scale that can be achieved with large systems in terms of energy, labour and sludge management. Conversely, conventional electro-mechanical wastewater treatment plants (WWTPs) that serve small decentralised agglomerations are at the lower end of available scale economies, which results in higher initial capital (CAPEX) and operational costs (OPEX) per capita [1]. Geographical remoteness can lead to problems with suppliers and services, and in some cases the lack of experienced plant operators and managerial staff can reduce the options for the type of system that can be practically implemented. All of these factors make the task of system selection both difficult and economically critical. According to Molinos-Senante et al. [2] the selection of the most appropriate wastewater treatment technology is the biggest challenge faced by wastewater treatment management. It is, therefore, imperative that wastewater treatment project commissioners understand how alternative systems will perform economically under different site-specific conditions, and have the tools with which to do so.

Several approaches have been proposed to address the issue of site-specific conditions in the wastewater treatment system (WWTS) selection process. The application of the multiple attribute decision-making (MADM) method to wastewater treatment was originally adopted by Tecle et al. [3]. The approach was to carry out system assessments based on

* Corresponding author.

Presented at the 13th IWA Specialized Conference on Small Water and Wastewater Systems & 5th IWA Specialized Conference on Resources-Oriented Sanitation, 14 –16 September 2016, Athens, Greece.

a set of criteria such as the level of influent pollution and required effluent quality. However, CAPEX and OPEX factors were treated individually and not considered from a life cycle perspective. An analytical hierarchy process was developed by Ellis and Tang [4]. In this method, a hierarchy model for system selection was developed with data gathered from existing plants. The parameters used in the study included many of the quantitative parameters used by Tecle et al. [3], but also included several subjective, qualitative criteria such as "ability of local administration to adequately support the work's operation" and "willingness and enthusiasm of community/politicians to improve the existing wastewater treatment facilities." Qualitative parameters of this nature can be difficult to assess. The weighting of these types of criteria is opinion based and can be subject to small temporal variations. Public opinion can change quickly in reaction to a negative event such as a water contamination or a bathing restriction. The absence of a sludge treatment/disposal criterion is significant because of the effect that sludge treatment/disposal can have on the economic performance of a WWTP. An innovative approach was developed by Kalbar et al. [5] to address system selection in India. The methodology presented is MADM based; however, unlike the aforementioned methodologies that use a list of criteria, this method includes the six specific scenarios most commonly found in India. Each scenario has three levels of information. The first level defines the location type: urban, sub-urban and rural. The next level provides a choice between locations with and without land restrictions, and lastly between systems that discharge to a waterbody and systems that require water reuse. The six scenarios are then evaluated with a set of weighted criteria. The criteria include life cycle costs presented as net present worth, land requirement and environmental impact is accounted for with global warming and eutrophication inventories. There are also a number of qualitative criteria such as reliability, durability and acceptability.

The system selection methods reviewed in the previous paragraph consider whole life cycle costs that include many externalities and indirect costs. This paper presents a methodology for assessing and comparing direct costs of alternative treatment systems for user-defined site-specific scenarios. The scope of the project relates to small WWTPs (500–2,000 PE [population equivalent]) in Ireland, but the approach has universal application. A decision support tool (DST) has been designed and developed that allows the user to input multi-parameter, site-specific data such as influent loading, discharge limits (DLs), footprint and sludge treatment option; the tool then outputs CAPEX, OPEX and life cycle costs information based on user inputs. Results are presented for several test scenarios that vary in scale, loading and DLs.

2. Background

The proposed location for a WWTP will ultimately determine the most economically feasible system for a specific site. WWTP capital and operational expenditure is influenced by site-specific factors. Scale, loading, DLs, land availability, sludge management alternatives and available labour are site-specific factors that will affect the economic performance of a WWTP. Different wastewater treatment systems will have a different operational cost distribution in a given location under a given set of conditions. It is, therefore, important to understand how different wastewater treatment systems perform under certain conditions in order to select the most appropriate system for a particular location. The following sections discuss some of the key factors that influence WWTP economic performance.

2.1. Discharge limits

DLs could be considered one of the main deciding factors in the system selection process. On the 21st of May 1991, the then European Economic Community (EEC) issued the 91/271/EEC Urban Wastewater Treatment Directive (UWWTD) [6]. The DLs shown below (Tables 1 and 2) were intended for agglomerations over 2,000 PE. However, many local authorities in Ireland use these limits as the minimum standard for agglomerations below 2,000 PE. While most WWTSs can achieve the levels of substrate removal required by the assigned DLs, the costs associated with achieving these limits can vary significantly between systems. An example of a cost trade-off can be seen in the case where a low phosphorus limit is imposed on a plant. An anaerobic/anoxic/oxic (AAO) system will incur less chemical costs than an anoxic/oxic (AO) system due to the ability of the AAO systems to facilitate enhanced biological phosphorus removal (EBPR). However, the AAO systems will require additional initial capital investment for the anaerobic tank, and will incur an additional energy cost for mixing and pumping. The extent of the cost trade-off in this case will depend on the magnitude of phosphorus loading, the value of the phosphorus DL and the perceived lifetime of both systems as this will dictate the payback time for the additional capital investment in the AAO system. This is just one example of the many OPEX/CAPEX trade-offs that can occur between competing systems.

Table 1
Regulations concerning discharge from urban wastewater treatment plants

Parameter	Concentration	% Removal
BOD_5 (mg O_2/L)	25	70–90
COD (mg O_2/L)	125	75
TSS (mg/L) (>10,000)	35	90
TSS (mg/L) (10,000 > PE > 2,000)	60	70

Table 2
Nutrient discharge limitations for sensitive areas

Parameter	Concentration	% Removal
Total phosphorous (mg/L) (10^5 > PE > 10^4)	2	80
Total phosphorous (mg/L) (>10^5)	1	
Total nitrogen (mg/L) (10^5 > PE > 10^4)	15	70–80
Total nitrogen (mg/L) (>10^5)	10	

2.2. Labour

Depending on the region or country, there may be issues recruiting necessary labour for small WWTPs. The lack of experienced personnel can have an adverse effect on OPEX, because properly trained and skilled personnel are essential for WWTP operational efficiency [7]. In a study carried out by Hegg et al. [8], 30 WWTPs were evaluated to determine the factors affecting plant performance. It was found that the top two factors limiting performance were:

- operator application of concepts and testing to process control and
- wastewater treatment understanding.

The specific cost of labour will vary with location, and in many cases the percentage of OPEX attributable to labour is significantly higher for small WWTPs [9]. Kemper et al. [10] reported that the ratio of labour costs to overall OPEX is much lower in EU countries when compared with some developing countries (Fig. 1). However, it is difficult to ascertain the exact causes of these large differences from the presented data. It is likely that there are a number of reasons rather than one single attributable factor. There may be a scarcity of local experienced wastewater treatment personnel, forcing authorities to provide higher salaries to attract professionals from outside the area. The degree of system automation may be higher in some EU states, requiring less human input. It may also be the case that some of the other operational costs such as sludge disposal, energy and chemicals are lower in these regions.

2.3. Sludge management

Values in the literature for sludge transport and disposal costs vary between €100 and €200 per tonne of dry solids (DS) depending on the final destination of the sludge (agriculture, composting and incineration) [11], and can account for between 15% and 20% of the total operational cost. Figures reported by the European Commission suggest that between 2006 and 2009 more than 10 million tonnes DS were produced by the 27 EU member states [11]. This figure is expected to rise both as a result of general population increase and the continued implementation of the UWWTD (91/271/EC) [6]. Land filling has been used extensively across Europe and has historically been the most cost-effective method of sludge disposal. However, since the introduction of the EC Landfill Directive (1999/31/EC) [12] there has been a sharp decline in landfilling of sludge, and in some EU states such as Germany [13], the practice has been banned completely unless the sludge is in the form of ash from sludge incineration. There are also growing concerns over continued application of sludge to farmland. Farmers in countries such as Sweden have taken it upon themselves to stop the practice completely [14]. The result of an increase in sludge volume and more stringent sludge disposal regulations means that the cost of sludge disposal has seen a substantial increase. Small WWTPs in particular suffer a double economic blow in relation to sludge management when compared with large centralised systems. It is not economically feasible for plants below agglomeration sizes of 40,000 PE to employ anaerobic digestion [15] and, therefore, they do not benefit from an energy return. There will also be additional costs to stabilise sludge through other means. Furthermore, the costs associated with sludge dewatering may not be economically feasible, resulting in large volumes of sludge to be transported off-site.

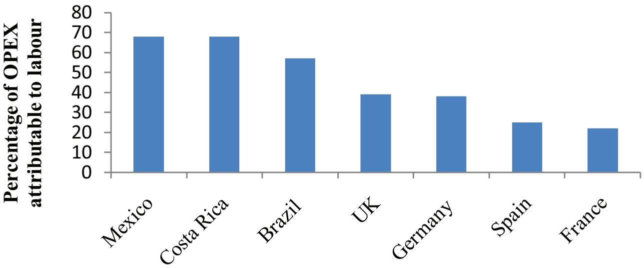

Fig. 1. Labour cost to OPEX ratio (adapted from [7]).

2.4. Land availability and specific location

The location of a plant affects a range of capital and operational costs. Depending on the scale of a WWTP, the cost of land in a particular location may be such that the plant footprint becomes a factor during the selection process. However, it is more likely that issues relating to land will revolve around availability rather than cost. Space restrictions may exclude some natural systems as an option, or even some of the larger electro-mechanical systems such as extended aeration. The cost of civil works can be affected by topography and soil condition. Proximity to residential areas can add to capital costs if strict odour control is required, necessitating process covers or buildings, odour scrubbing towers, VO_2 monitors and odour extraction systems. In some cases, authorities may face lengthy legal challenges during the planning process from resident associations and other interest groups; however, public opposition of this nature tends to be directed against the disruption caused during the construction of large-scale systems rather than their operation. The distance to suppliers will dictate both the cost of delivery and the amount of storage required for chemicals and other materials.

3. Life cycle costing

Economic considerations during the system selection process can sometimes be dominated by the CAPEX [16], with secondary consideration given to operation and maintenance costs. It is widely accepted that this approach is misguided and lacks transparency because in many situations the cost of acquisition of a system can be small in relation to the cost of ownership [17]. Life cycle costing (LCC) [18] is a holistic approach that is used to assess the economic feasibility of a system over the entirety of its predicted lifetime. The LCC methodology provides a comprehensive assessment of costs and trade-offs between competing alternatives by accounting for both CAPEX and projected OPEX over the lifetime of a product, service or system [19]. The net present value (NPV) method is commonly used to calculate life cycle costs (Eq. (1)):

$$\text{NPV} = \text{Initial Cost} + \sum_{k=1}^{N} \text{Future Cost}_k \left[\frac{1}{(1+d)^{n_k}} \right] \tag{1}$$

where the initial cost is the CAPEX in year 0, n is the year of expenditure, k is the item of expenditure and d is the discount rate in year n. The future cost expression on the right-hand side of the equation accounts for all future operation and maintenance costs. This expression can be further divided into single and recurring future costs as per the approach of Rawal and Duggal [20]. Their approach was to use the single present value (SPV) formula (Eq. (2)) for one-off cash flows such as the CAPEX, capital replacement and the residual value at the end of the system lifetime, and the uniform present value (UPV) formula for annually recurring operational costs (Eq. (3)), where A_0 is the annual recurring cost at year 0. Therefore, the total system NPV used in this study is given by Eq. (4):

$$\mathrm{SPV} = \frac{FV}{\left(1+d\right)^n} \tag{2}$$

$$\mathrm{UPV} = A_O \frac{\left(1+d\right)^n - 1}{d\left(1+d\right)^n} \tag{3}$$

$$\mathrm{NPV} = \mathrm{SPV} + \mathrm{UPV} \tag{4}$$

4. Methodology

A DST was developed on the Microsoft Excel VBA platform. A selection of six WWTSs was chosen for the study. These include five electro-mechanical systems and one natural system (Table 3). The electro-mechanical systems include inlet works, primary sedimentation, biological treatment, secondary sedimentation and optional sludge treatment. It should be made clear at this point that the purpose of this paper is to present a methodology and illustrate the effects that scenario variation can have on a system's economic performance. It is not intended that the results be used as a definitive measure of a systems economic feasibility, as many of the specific costs used in the study can vary significantly depending on location.

Toolkit user input is limited to loading, DLs, sludge treatment and disposal option and surface area limit (if applicable). Inputs common to loading and DLs are biochemical oxygen demand (BOD), chemical oxygen demand (COD), total suspended solids (TSS), total nitrogen (TN), total phosphorus (TP), ammonia (NH_3), and orthophosphate (PO_4^{3-}). Additional DLs include options for chlorination and dechlorination. Plant scale can be entered either as agglomeration size or hydraulic load if it is known.

The toolkit outputs are NPV, CAPEX, OPEX, energy use, sludge production and surface area (Fig. 2). An OPEX distribution profile is presented for each system giving details of energy, sludge disposal, chemical costs and labour (Fig. 3). Further breakdown of energy use distribution and chemical cost distribution are also included in the toolkit outputs. Details of calculation methods are discussed in the following section.

4.1. Cost calculations

Values for CAPEX are limited to variations in scale only. Power law models were developed from compiled data and normalised to an Irish context (Eq. 5) [21,22]:

$$\mathrm{C}_c = \left(\frac{I_c C_t K_l}{I_t}\right) \times \mathrm{ER}_l \tag{5}$$

where C_c is the current cost of the system, I_c is the current construction cost index, I_t is the construction cost index at time t of plant construction, C_t is the cost of construction at time t, K_l is the location factor (Ireland – United States location

Table 3
Wastewater treatment systems included in the study

Suspended growth	Attached growth	Hybrid	Natural
Anaerobic/anoxic/ oxic (AAO)	Rotating biological contactors (RBC)	Integrated fixed-film activated sludge (IFAS)	Constructed wetlands (CW)
Anoxic/oxic (AO)	Trickling filters (TF)		

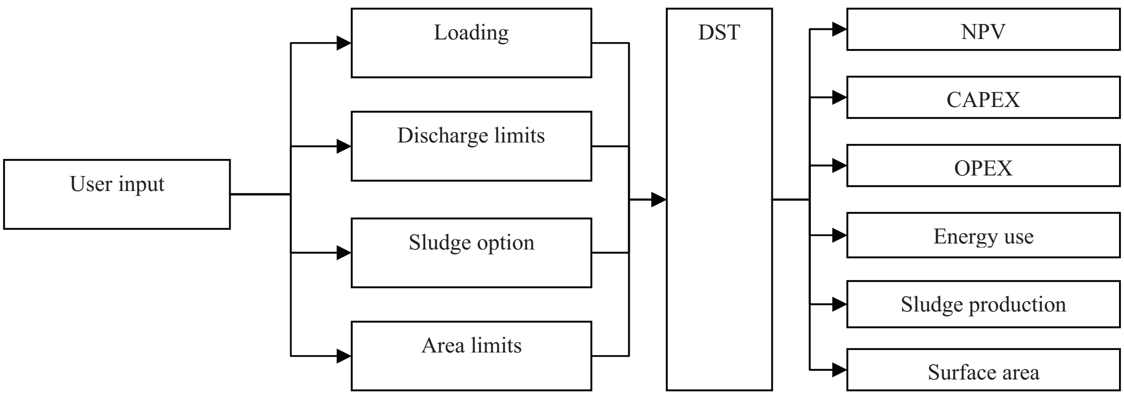

Fig. 2. Decision support tool flow schematic.

factor 2015 = 1.3), ER_1 is the currency exchange rate (€ – U.S.$, in 2015 ≅ 0.9). The CAPEX for each system includes the cost of engineering, civil works, electro-mechanical equipment for inlet works, primary and secondary treatment, sludge dewatering, chlorination and 15% contingency. Replacement capital has not been included in the LCC calculations as this requires system specific data that were not available at the time of the study. However, it is advised that replacement capital be included whenever possible as this can be a significant cost factor in certain systems. It is assumed that the plant sizes are too small to generate any direct revenue from supplying electricity back to the grid or the sale of biosolids. Life cycle costs are calculated using the NPV formula presented (Eq. (4)). A commonly adopted discount rate in the literature is 3.5% [20]. The Irish National Development Finance Agency currently recommend using a nominal discount rate of 3.96%

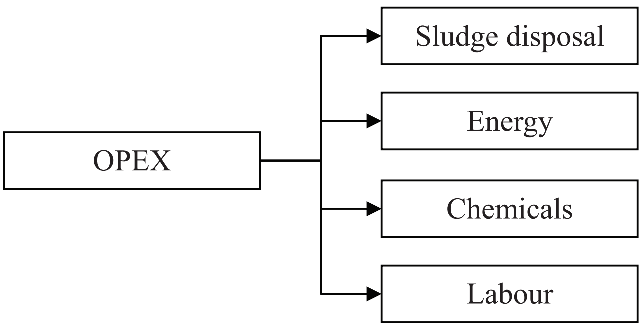

Fig. 3. Operation expenditure distribution.

for project lifetimes of between 10 and 20 years, and suggest a 5% test discount rate for use in cost benefit analysis [23]. A real discount rate of 3.5% is used in this study, but the parameter has been soft coded into the toolkit to accommodate variance. The systems lifetime is set at 25 years. The recurring OPEX components are energy, labour, sludge disposal and chemicals.

Energy modelling includes aeration energy for activated sludge systems and pumping. For other unit processes such as RBC motors, sludge dewatering plant, primary and secondary settling, and inlet works, average power requirement values from the literature and from manufacturers design specifications have been used. Municipal energy such as that used for lighting, utilities and control has been given an average value of 2% of the overall energy used by the plant [24].

RBC system energy requirements are dominated by the power required for shaft rotation. Shaft rotation energy demand is calculated as a function of the required disc surface area. A linear regression model was developed based on the study carried out by Gilbert et al. [25], and is given by Eq. (6):

$$E_{\mathrm{RBC}} = (184.382 \times 10^{-6}) A_{\mathrm{disc,req.}} \tag{6}$$

where E_{RBC} = energy required (kWh/m^2); $A_{\mathrm{disc,req}}$ = disc area required (m^2).

Labour calculations are based on the methodology proposed by the New England Interstate Water Pollution Control Commission [32]. The labour categories, description and specific cost are presented below in Table 4.

Table 4
Energy and related system parameters, reported value ranges and assumed values

Parameter	Variation/range	Assumed values	Source
Aerator system	Submerged diffuser		
Diffuser type	Fine bubble		
Oxygen transfer efficiency	Range (kg O_2/kWh)	3.5	[26,27]
• Fine bubble diffusers	3.0–4.8		
Alpha factor (α)	Variable	Function of SRT (Figs. 5–7)	[28]
• Fine bubble diffusers			
Beta factor (β)	0.97–0.99	0.9	[29]
Fouling factor	0.4–1	0.9	[30]
Tank depth (m)	4–6	Variable based on tank surface area to depth ratio	
Tank shape	Rectangular		
Blower efficiency	0.45–0.65	0.60	[24]
Motor efficiency	0.85–0.95	0.90	[24]
Pump efficiencies		0.55	[24]
Temperature (°C)	Variable	10	
Elevation (meters above sea level)	Variable	118	
RBC motor energy (kWh/m^2)		see Eq. (6)	[25]
Volute energy (kWh/kg DS)		0.05	[31]
Inlet works energy (kWh/m^3)		0.01	[24]
Clarifier energy (kWh/m^3)		0.012	[24]

Sludge quantities are based on loading and DL values input by the user. Primary, secondary, and attached growth sludge DS concentrations are assumed to be 5%, 1.8% and 2.3%, respectively [33]. There are two sludge management options: (1) invest in sludge treatment plant (in this case a volute thickener/dewatering unit, delivering DS concentrations of 24%), and incur additional OPEX for energy and chemicals with final disposal by external contractor; or (2) completely outsource disposal of untreated sludge to external contractor. A conservative value of €20/m^3 is used in this study. Additional sludge treatment plant options can be included (centrifuges, belt thickeners, etc.) which will yield varied sludge volumes. Also, the form of ultimate disposal could include delivery to a parent plant if this option is available. For the purpose of demonstration, the choice is limited to either volute treatment and disposal, or complete outsourcing.

4.2. Scenarios

24 scenarios were assessed across the six treatment systems. These include variations in agglomeration, DLs and sludge treatment. The agglomeration scale varies from 500 to 2,000 PE. Medium strength loading is applied as per the definition given by [26], and presented in Table 5. Variations in DLs are presented in Table 6. The inclusion or omission of disinfection would not affect comparative assessment as disinfection cost calculations are a function of influent flow rate which is equal for all system. Moreover, the values presented are not intended to be absolute due to the highly variable nature of specific costs. Thus, disinfection is not included as a requirement for the scenarios presented here. The scenario descriptions are presented in Table 7.

Table 5
Labour categorisation, description and cost

Labour type	Description	Cost per hour (€/h)
Operator	General operation	20
Engineer	Carries out technical maintenance, operation and trouble shooting	28
Lab technician	Carries out water quality analysis	20
Yard hand	Carries out low-level janitorial tasks such as grass mowing, painting, rust removal	12

Table 6
Discharge limits and loading

	Loading	High DL	Medium DL	Low DL
BOD, mg/L	350	35	25	15
COD, mg/L	750	125	100	80
TSS, mg/L	400	35	30	25
TN, mg/L	60	20	15	10
TP, mg/L	15	2	1.5	1
NH_3, mg/L	45	1	0.75	0.5
PO_4^{3-}, mg/L	10	1	0.5	0.1

5. Discussion

5.1. Present value assessment

Fig. 4 illustrates a general trend occurring across all scenarios whereby the constructed wetland (CW) NPV is significantly less than those of conventional electro-mechanical systems. This is a common finding in studies of a similar nature [20,34]. CW NPV is least affected by variations in DLs and the inclusion or omission of sludge treatment. A variation in CW NPV is generally attributable to differences in CAPEX, because the DLs will dictate the type, or combination of CWs required for a particular location at the design stage. Therefore, the remainder of the discussion will focus on electro-mechanical systems.

In most scenarios, the trickling filters (TF) system had the highest NPV except for S1 and S4 (Fig. 5) where the integrated fixed-film activated sludge NPV was 2.5% and 4.8% higher, respectively. This can be attributed to large TF CAPEX. This is further evident in scenarios where sludge treatment is included (Figs. 6, 8–10). The ratio of OPEX to CAPEX is much smaller for TF systems than for others, and the introduction of sludge treatment has a less influence on TF NPV. The difference between TF NPV in S1 and S5 (Fig. 5) is less than 3%; compared with an almost 16.6% reduction in AAO NPV for

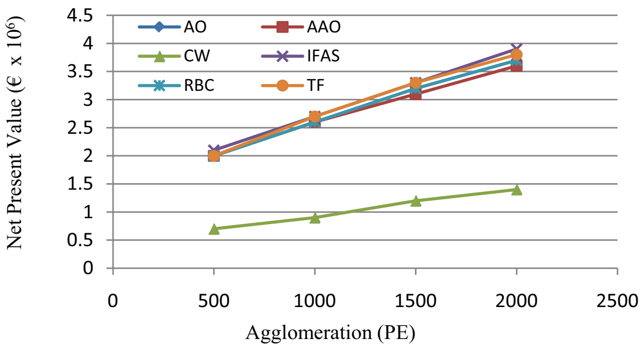

Fig. 4. General NPV trend showing large difference between natural and conventional electro-mechanical systems.

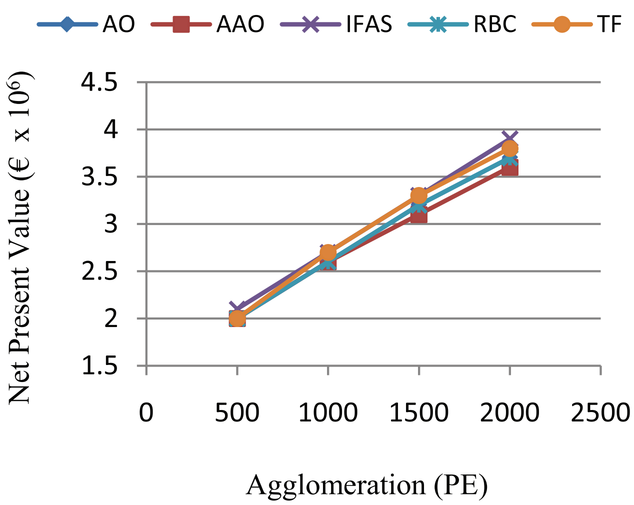

Fig. 5. Scenarios 1–4.

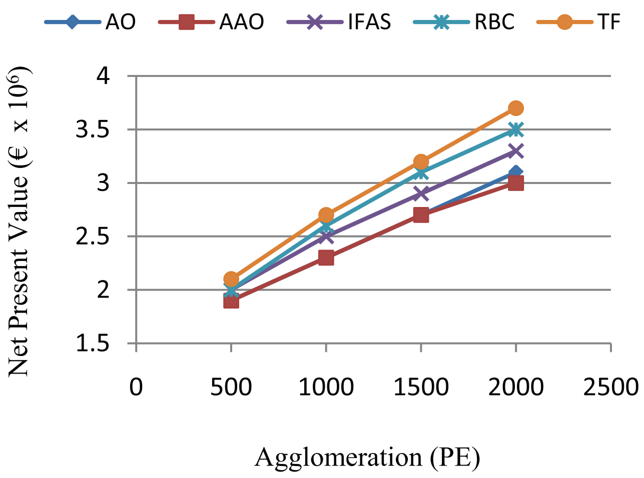

Fig. 6. Scenarios 5–8.

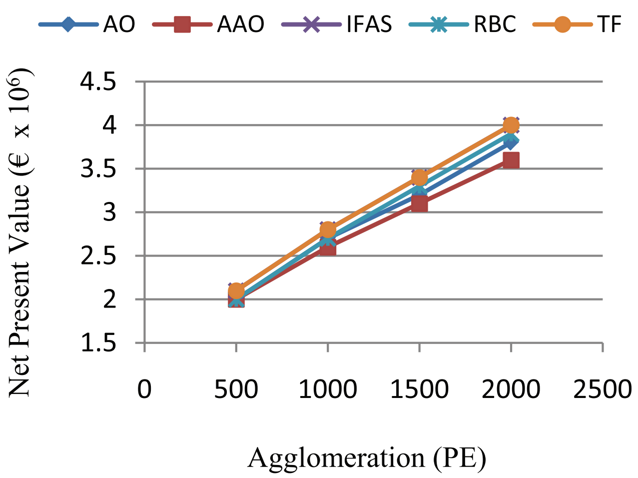

Fig. 7. Scenarios 9–12.

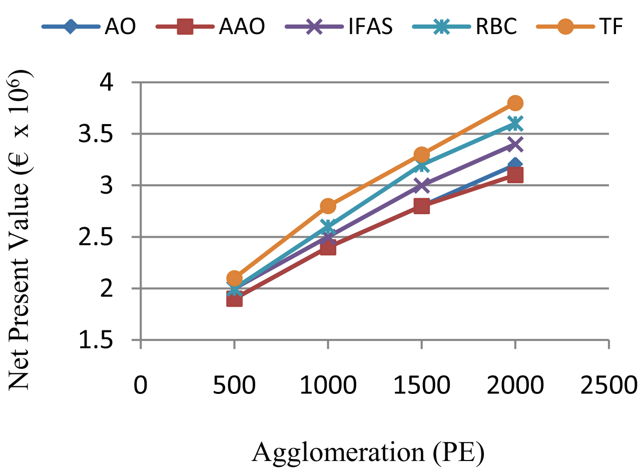

Fig. 8. Scenarios 13–16.

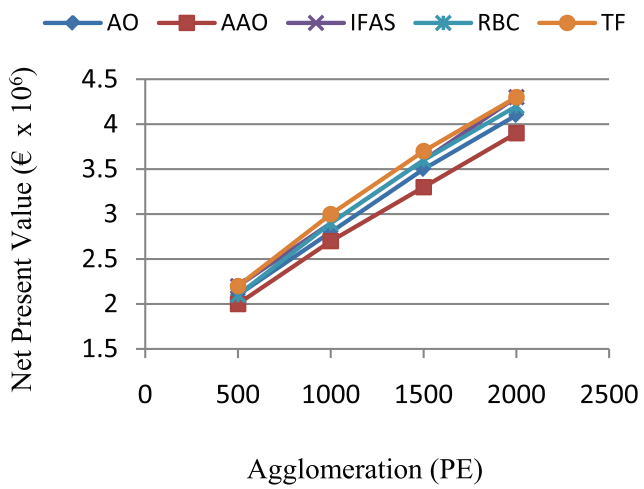

Fig. 9. Scenarios 17–20.

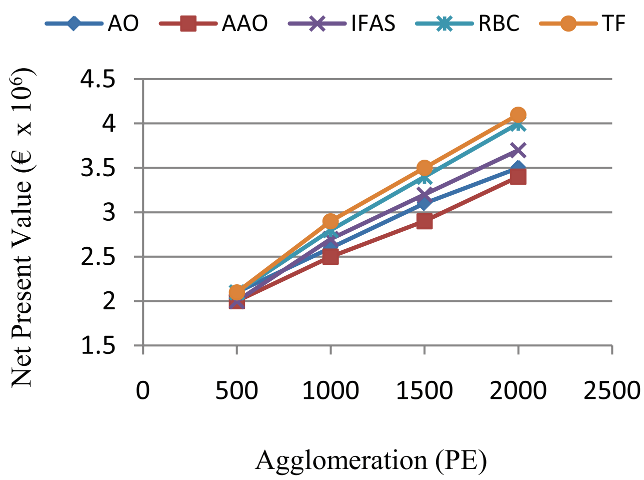

Fig. 10. Scenarios 21–24.

the same scenarios. The best economic performance exhibited by the TF system was in S1 where it had the joint lowest NPV with the AO, AAO and RBC systems.

The AAO system had the lowest NPV in all scenarios, and was the least affected by reductions in DLs. This can be attributed mostly to the lower demand for phosphorus precipitating chemicals. The reduction in DLs had the largest effect on the RBC system which exhibited a 12.5% increase in NPV at 2,000 PE. This is due to the large increase in growth media area with respect to increases in ammonia reduction requirements.

The inclusion of sludge treatment is most significant with larger agglomerations. The largest reduction in NPV occurs with the AAO system at 2,000 PE at the highest DLs. At 500 PE most systems exhibit very little change in NPV with respect to the inclusion of sludge treatment, with an average of 2.5% across all scenarios. Moreover, there is a negative reduction of cost, that is, an increase in NPV with TF systems at the highest DLs. At this agglomeration scale, the economic balance between the cost of investment and ownership of sludge treatment plant, and the gains from reductions in sludge volume begins to tip towards complete out-sourcing of sludge treatment and disposal to an external contractor. However, it should be noted that the value used for sludge disposal by contractor was conservative (€20/m^3).

5.2. Operational costs

Figs. 11–14 present the OPEX profiles for the AAO and TF systems at high and low DLs. These systems have been chosen because they generally had the lowest and highest NPVs, and were the most and least sensitive to scenario variation. There are two points of note to consider before examining the OPEX distribution profiles of these systems: first, the energy values produced by the toolkit are ideal values

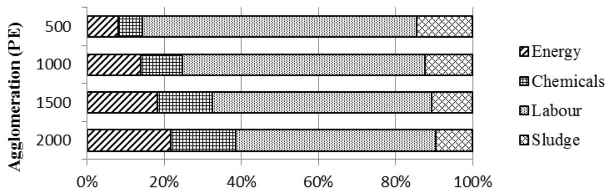

Fig. 11. OPEX distribution for AAO system with high discharge limits.

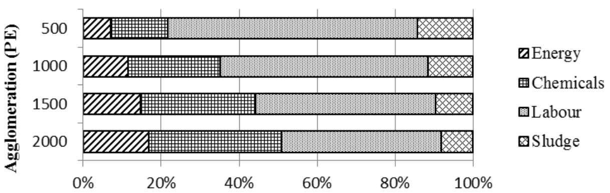

Fig. 12. OPEX distribution for AAO system with low discharge limits.

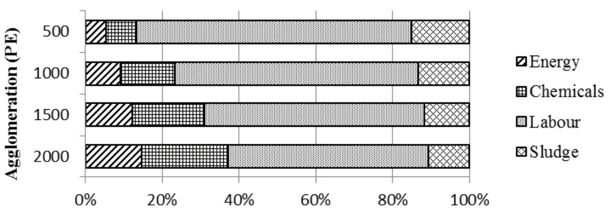

Fig. 13. OPEX distribution for TF system with high discharge limits.

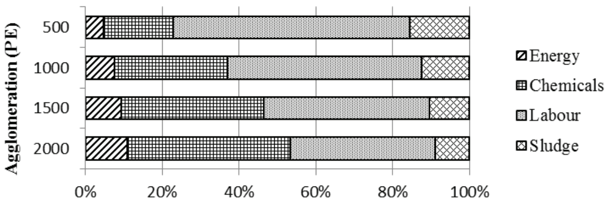

Fig. 14. OPEX distribution for TF system with low discharge limits.

developed from first principles and are generally low when compared with actual plant energy data. There are many areas of a treatment system that contribute to energy loss which are difficult to account for such as poor motor and pump efficiencies, alpha and beta factors in aeration systems, diffuser head fouling, various sources of head loss throughout the pipe network, and poor operational and maintenance practice. There are also a variety of energy scale economies that while visibly present are often difficult to capture and model. In reality, energy values will be higher and therefore, the energy cost percentages presented here should be viewed as benchmark values. The second point of note is in relation to the percentage contribution of sludge disposal costs. The values presented here account only for the cost of removal of sludge from the plant site. Other costs related to the treatment and disposal of sludge are included in other cost elements. Handling costs are included in the labour values, and dewatering costs are included in the energy and chemical costs.

5.3. Operational cost calculations

A detailed breakdown of OPEX calculations for the 2,000 PE AAO system (Fig. 11) is provided below as an example (Tables 8–12).

The OPEX distribution is dominated by labour costs in all scenarios. The percentage contribution to labour increases

Table 7
Scenario descriptions

Scenario	Agglomeration scale (PE)	Discharge limits	Sludge treatment
S1	2,000	High	x
S2	1,500	High	x
S3	1,000	High	x
S4	500	High	x
S5	2,000	High	✓
S6	1,500	High	✓
S7	1,000	High	✓
S8	500	High	✓
S9	2,000	Medium	x
S10	1,500	Medium	x
S11	1,000	Medium	x
S12	500	Medium	x
S13	2,000	Medium	✓
S14	1,500	Medium	✓
S15	1,000	Medium	✓
S16	500	Medium	✓
S17	2,000	Low	x
S18	1,500	Low	x
S19	1,000	Low	x
S20	500	Low	x
S21	2,000	Low	✓
S22	1,500	Low	✓
S23	1,000	Low	✓
S24	500	Low	✓

Table 8
Labour hours distribution for the 2,000 PE AAO system

Operator	h/y	Maintenance	h/y
Preliminary	119.52	Screen cleaning	65
Primary settling	100	Chemical addition	26
Activated sludge	401.48	Clarifiers	65
Nitrification	42.31	Pumps	65
EBPR	42.31	Blowers	52
Chemical P removal	27.51	Chlorination	0
Chlorination	0	Dechlorination	0
Dechlorination	0	Instrumentation calibration	26
Sludge handling	65	Mechanical mixers	26
Sludge drying beds	0	Total annual	325
Total annual	798.13	Hours per day	1.25
Hours per day	3.07	*Yard hand*	
Lab technician		Janitorial	100
Testing	34.5	Mowing	60
Per day	0.13	Painting	60
		Rust removal	60
		Total	280
		Hours per day (@260/year)	1.08

Table 9
Labour cost calculation

	h/d	Cost (€/h)	Cost (€/d)
Operator	3.07	20	61.4
Lab technician	0.13	20	2.6
Maintenance	1.25	28	35
Yard hand	1.08	12	12.96
Total cost per day			111.96

as the plant size decreases. This is most prominent in the high DL scenarios (Figs. 11 and 13). Labour accounts for over 70% of OPEX in both systems at 500 PE and just over 50% at 2,000 PE. In 500 PE low DLs scenarios, the labour percentages are reduced to 64% and 61% for the AAO and TF systems, respectively. The difference here can be attributed in part to the increase in the percentage contribution of chemical costs in the TF system ($FeCl_3$ – 35 L/d compared with 27 L/d in the AAO system), but also to the difference in energy demand – 0.43 and 0.26 kWh/m^3 for the AAO and TF systems, respectively. The large differences in the energy values presented here reflect similar findings reported by Burton [35]. The chemical cost percentage is the highest operational cost element in the TF system at 2,000 PE, which accounts for just over 42% of operational expenditure. This figure is reduced to 38% when all sludge costs are aggregated in a single value (Fig. 15).

Table 10
Sludge disposal cost calculations

Sludge type	(kg/d)	Dry solids (%)	Volume (m^3/d)
Primary sludge (kg/d)	112.76	5.00	2.26
WAS (kg/d)	60.17	1.80	3.34
Additional sludge from P removal (kg/d)	23.78	5.00	0.48
Mixed sludge (kg/d)	196.71	3.24	6.07
Additional solids production from dewatering (kg/d)	49.18		
dewatered sludge (kg/d)	245.89	24.00	1.02
Disposal cost per day	Cost (€/m^3)	Volume (m^3)	Cost (€/d)
	20	1.02	20.4

Table 11
Energy cost calculations

Energy (kWh/d)	
Aeration energy	120.62
Inlet works	4
Pumping energy	36.61
Primary settling	4.8
Secondary settling	4.8
Sludge dewatering and thickening	9.10
Mixing energy	1.03
Municipal energy	3.60
Total (kWh/d)	184.55
Cost (@ €0.25/kWh) (€/d)	46.14

Table 12
Chemical cost calculations

Chemicals	Specific cost	Quantity	Cost per day
$FeCl_3$ (€/L)	0.7	39.95 L	27.96
SO_2 (€/kg)	0.5	–	0.00
$Ca(OH)_2$ (€/kg)	0.2	39.34 kg	7.87
$Ca(OCl)_2$ (€/kg)	0.8	–	0.00
CH_3OH (€/L)	0.7	–	0.00
Total			35.83

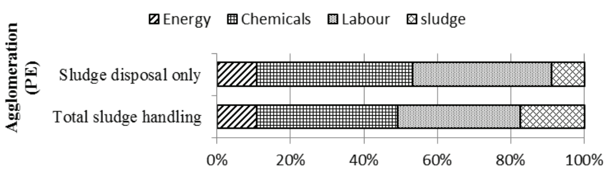

Fig. 15. Trickling filter OPEX at 2,000 PE with low DLs.

6. Conclusions

The primary objective of this study was to illustrate the dynamics of cost distribution with respect to changes in conditions, and to demonstrate the advantage of a scenario-specific system assessment tool. The study has shown that there is no "one size fits all" wastewater treatment design solution for small systems. Variations in plant scale, organic loading, DLs and land availability can influence the economic performance of a treatment system to the extent that its suitability for a given location may be less than that of competing systems, and therefore, system applicability should be assessed on a case by case basis.

CWs proved economically to be the best treatment option for locations where land availability is not an issue. The low energy, material and labour input makes CW systems ideally suited to small rural and isolated locations.

Energy use contributes significantly to the operational cost of small electro-mechanical wastewater treatment systems. Small agglomerations are at the negative end of energy scale economies and incur a higher per capita energy cost. This places an even greater importance on understanding the specific energy costs associated with a given system in a given location.

Aeration delivery and pumping are the primary energy sinks in suspended growth systems and can collectively account for up to 90% of total energy use. Attached growth systems have lower energy consumption in BOD removal only scenarios; however, their capacity for nutrient removal is less than that of suspended growth systems and requires additional energy and material inputs, and may in some cases require a tertiary process to achieve very low final effluent nutrient concentrations.

The use of a scenario-specific DST during the wastewater treatment system selection process can provide a better understanding of the economic performance of individual systems and processes. The support tool provides users with a platform to assess the variety of trade-offs that can occur between a system's capital and operational expenditure in different scenarios.

7. Further work

All of the costs that are presented in this study are costs directly associated with the treatment of wastewater. However, there are other elements of cost that are not included in this toolkit. Externalities such as social benefits and environmental impact have indirect contributions to cost and should be accounted for. Future additions to the toolkit will include a life cycle assessment component to account for environmental impact and elucidate the economic–environmental trade-offs that exist both within, and between systems.

Acknowledgements

This study has been conducted as part of the EPA Research Programme 2014–2020. The programme is financed by the Irish Government. It is administered on behalf of the Department of the Environment, Community and Local Government (DECLG) by the EPA, which has the statutory function of co-ordinating and promoting environmental research.

References

[1] M. Boller, Small wastewater treatment plants: a challenge to wastewater engineers, Water Sci. Technol., 35 (1997) 1–12.
[2] M. Molinos-Senante, T. Gómez, R. Caballero, F. Hernández-Sancho, R. Sala-Garrido, Assessment of wastewater treatment alternatives for small communities: an analytic network process approach, Sci. Total Environ., 532 (2015) 676–687.
[3] A. Tecle, M. Fogel, L. Duckstein, Multicriterion selection of wastewater management alternatives, J. Water Resour. Plann. Manage., 114 (1988) 383–398.
[4] K. Ellis, S. Tang, Wastewater treatment optimization model for developing world. I: Model development, J. Environ. Eng., 117 (1991) 501–518.
[5] P.P. Kalbar, S. Karmakar, S.R. Asolekar, Selection of an appropriate wastewater treatment technology: a scenario-based multiple-attribute decision-making approach, J. Environ. Manage., 113 (2012) 158–169.
[6] European Council, Directive: 91/271/EEC concerning urban waste water treatment, Off. J. Eur. Union, 135 (1991) 40–52.
[7] R.L. Michel, A.L. Pelmoter, R.C. Palange, Operation and maintenance of municipal waste treatment plants, J. Water Pollut. Control Fed., 41 (1969) 335–354.
[8] B.A. Hegg, K.L. Rakness, J.R. Schultz, Evaluation of operation and maintenance factors limiting municipal wastewater treatment plant performance, J. Water Pollut. Control Fed., 50 (1978) 419–426.
[9] E. Reicherter, Investigations to Indicators as a Basis for Cost Considerations in Sanitation (in German), Oldenbourg-Industrieverl., Munich, Germany, 2003.
[10] K. Kemper, G. Yepes, M. Garn, Personnel Costs as an Indicator for W&S Utility Performance in Developing Countries, The World Bank: Transportation, Water and Urban Development Department, Infrastructure Note WS-12, Washington D.C., 1994.
[11] G. Goldenman, J. Middleton, Environmental, Economic and Social Impacts of the Use of Sewage Sludge on Land, EU Commission, Brussels, Belgium, 2008.
[12] European Council, Directive 1999/31/EC on the landfill of waste, Off. J. Eur. Union, 182 (1999) 1–19.
[13] R.J. LeBlanc, P. Matthews, R.P. Richard, Global Atlas of Excreta, Wastewater Sludge, and Biosolids Management: Moving Forward the Sustainable and Welcome Uses of a Global Resource, UN-Habitat, 2009.
[14] B. Hultman, E. Levlin, K. Stark, Swedish Debate on Sludge Handling, Proc. Polish–Swedish Seminar on Sustainable Municipal Sludge and Solid Waste Handling, Cracow, Poland, 2000.
[15] S. Malamis, E. Katsou, F. Fatone, Integration of Energy Efficient Processes in Carbon and Nutrient Removal from Sewage, Sewage Treatment Plants: Economic Evaluation of Innovative Technologies for Energy Efficiency, IWA Publishing, UK, 2015, p. 71.
[16] D.G. Woodward, Life cycle costing—theory, information acquisition and application, Int. J. Project Manage., 15 (1997) 335–344.
[17] I. Eisenberger, G. Lorden, Life-cycle Costing: Practical Considerations, DSN Progress Report 42~40, 1977, pp. 102–109.
[18] D. Arditi, H.M. Messiha, Life cycle cost analysis (LCCA) in municipal organizations, J. Infrastruct. Syst., 5 (1999) 1–10.
[19] B.S. Dhillon, Life Cycle Costing for Engineers, CRC Press, Boca Raton, United States, 2009.
[20] N. Rawal, S. Duggal, Life cycle costing assessment-based approach for selection of wastewater treatment units, Natl. Acad. Sci. Lett., 39 (2016) 103–107.
[21] G.W. Foess, P. Steinbrecher, K. Williams, G. Garrett, Cost and performance evaluation of BNR processes, Florida Water Resour. J., 11 (1998) 11–16.
[22] D. Gkika, G.D. Gikas ,V.A. Tsihrintzis, Construction and operation costs of constructed wetlands treating wastewater, Water Sci. Technol., 70 (2014) 803–810.
[23] National Development Finance Agency, 2017 Expenditure: Project Discount and Inflation Rates, 2016.

[24] Metcalf & Eddy Inc., Wastewater Engineering: Treatment and Resource Recovery, International ed., Vol. 1, McGraw-Hill, New York, 2014.
[25] W.G. Gilbert, J.F. Wheeler, A. MacGregor, Energy usage of rotating biological contactor facilities, J. Water Pollut. Control Fed., 58 (1986) 47–51.
[26] M. Henze, Biological Wastewater Treatment: Principles, Modelling and Design, IWA Publishing, UK, 2008.
[27] S. Bolles, Modelling Wastewater Aeration Systems to Discover Energy Savings Opportunities, Process Energy Services LLC, [Online] Available at: http://citeseerx.ist.psu.edu (Accessed 21 October 2015).
[28] D. Rosso, M.K. Stenstrom, Comparative economic analysis of the impacts of mean cell retention time and denitrification on aeration systems, Water Res., 39 (2005) 3773–3780.
[29] P.K. Tewari, J.K. Bewtra, Alpha and beta factors for domestic wastewater, J. Water Pollut. Control Fed., (1982) 1281–1287.
[30] M. Garrido-Baserba, R. Sobhani, P. Asvapathanagul, G.W. McCarthy, B.H. Olson, V. Odize, A. Al-Omari, S. Murthy, A. Nifong, J. Godwin, Modelling the link amongst fine-pore diffuser fouling, oxygen transfer efficiency, and aeration energy intensity, Water Res., 111 (2017) 127–139.
[31] Amcom. Volute Technology, [Online] Available at: http://en.amcon.co.jp/specvolute/#ec (Accessed 29 October 2016).
[32] NEIWPCC, The Northeast Guide for Staffing at Publicly and Privately Owned Wastewater Treatment Plants, NEIWPCC, Lowell, MA, United States, 2008.
[33] I.S. Turovskiy, P. Mathai, Wastewater Sludge Processing, John Wiley & Sons, New York, 2006.
[34] M.K. Gratziou, M. Tsalkatidou, N.E. Kotsovinos, Economic evaluation of small capacity sewage processing units, Global Nest J., 8 (2006) 52–60.
[35] F.L. Burton, Water and Wastewater Industries: Characteristics and Energy Management Opportunities, CEC Report 106941, Electrical Power Research Institute, St. Louis, MO, 1996.