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

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Abstract

Life-cycle costing models for 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 scale, loading, location and discharge limits. Capital and operational costs have been combined to produce life-cycle models for 9 treatment systems. Several scenarios with variations in scale, load and discharge limits were input to the support tool. The results show that in most scenarios, constructed wetlands represent the most economical option where surface area availability is not restricted. The percentage contribution of labour to operational costs increases as plant sizes are reduced.

Keywords

Wastewater treatment, capital and operational expenditure, life-cycle costs, organic loading, discharge limits.

INTRODUCTION

Small communities and rural agglomerations face a number of challenges in relation to their wastewater treatment requirements. Capital investment distribution can often favour large agglomerations where the potential risk of environmental and socio-economic consequence is higher. Smaller plants are at the lowest end of available scale economies, which makes operational costs higher per capita. Geographical isolation can lead to problems with suppliers and services, and in some cases a lack of experienced plant operators, engineers and managerial staff can reduce the list of available treatment options. However, much of the time, the main overriding issue is limited capital resources. This places a greater importance on the system selection process for small plants.

In wastewater treatment, the procurement process is often dominated by initial capital expenditure (CAPEX) requirements (Woodward 1997), with only secondary consideration given to operational expenditure (OPEX). It is widely accepted that this approach is flawed because in many situations the cost of acquisition of a system can be small in relation to the cost of ownership (Eisenberger, Lorden 1977). Life cycle costing (LCC) or life cycle cost analysis (LCCA) (Arditi, Messiha 1999) 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 more comprehensive and transparent assessment of costs and trade-offs between competing systems by accounting for both CAPEX and projected OPEX. A general LCC model is presented below (E.q.1) (Dhillon 2009).

$$LCC = RC + NRC \tag{1}$$

Where LCC is the item or system life cycle cost, RC is recurring cost (OPEX) and NRC is nonrecurring cost (CAPEX). Many industrial sectors have adopted the LCC approach, and LCC practitioners have developed sector specific LCC models.

CAPEX and OPEX estimations are commonly based on cost curves that are produced with aggregated data from a cohort of existing plants. Cost data are normally given as a function of design capacity in either €/ PE or €/m^3, and do not reflect the variety of different environmental conditions and legal requirements that an individual plant may be subject to. This is problematic because the environmental conditions under which a given plant has to operate can have a large effect on its economic performance, both from a capital and operational cost perspective. Therefore, it is important to develop as much as is practically possible, a scenario specific life cycle cost model that accounts for changes in conditions. The system selection process should consider factors such as land availability, geography, scale, topography, climate, discharge limits, distance to suppliers, sludge management and availability of qualified labour. Each of these factors to some degree will have an effect on the economic performance of a treatment system. Some of the key influencing factors are considered here in the following sections.

Discharge limits

Discharge limits can 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) (European Commission 1991). The discharge limits shown below (Table 1. and Table 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.

Most treatment systems can achieve high substrate removal efficiencies. However, some of the more stringent discharge limits can eliminate a system from consideration. For example, in the case of very low effluent suspended solid limits, systems with long solid retention times such as extended aeration can be prone to excessive effluent suspended solids concentrations if not properly monitored. This may necessitate the addition of a tertiary treatment stage, which will add to the already significant surface area requirements and subsequent operational costs. Ammonia discharge limits can affect a range of cost components. The additional area required by some systems can be as much as 1/3 of the total active surface area. Additional energy, chemicals and maintenance required for ammonia removal will contribute to an increase in operational costs.

Table 1. Regulations concerning discharge from urban wastewater treatment plants.

Parameter	*Concentration*	*Removal percentage*
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 percentage*
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	

Plant location

The location of a plant affects a range of capital and operational costs. Firstly, the cost of land in a particular location could be very high, resulting in plant footprint having to be considered. An urban location may have surface area restrictions eliminating large footprint systems such as constructed wetlands, lagoons, or 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 in some cases expensive odour extraction systems (although the latter is generally only applicable in much larger systems, and used in areas where strict emission limits are in place). The operational costs are also increased with the need for odour scrubbing chemicals and replacement media. The distance to suppliers can dictate the amount of chemical storage required, which, as well as increasing area requirements, can also drive up plant security and insurance costs.

Labour

Labour is the largest wastewater treatment operational cost and can account for between 30 and 40 % of the total OPEX for treatment plants below 10,000 PE. The hourly rate of operators, engineers and administrative personnel can vary significantly from country to country (Kampet 2001), but the labour-hours required can be considered consistent, and are a function of both scale of plant and system type. Several methods have been proposed to calculate labour-hours. Gratziou et al. (Gratziou, Tsalkatidou et al. 2006) proposed calculating labour-hours as a function of flowrate for both administrative and laboratory labour-hours. However, each system requires different levels of monitoring and control, and different levels of expertise. For example, extended aeration systems are ideally suited to rural isolate agglomerations because they are easy to operate and need minimal attention. Constructed wetlands require even less input and minimal expertise once properly designed. Contrary to these types of systems, IFAS systems have both suspended and attached growth processes occurring within the same system, and require more operator input, and a higher level of expertise.

Sludge disposal

The issue of sludge disposal is an area that requires particular attention from both an economic and environmental viewpoint. Figures reported by the European Commission suggest that between 2006 and 2009 more than 10 million tons DS (dry solids) were produced by the 27 EU member states (Goldenman, Middleton 2008). This figure is expected to rise both as a result of general population increase and the continued implementation of the Urban Wastewater Treatment Directive (91/271/EEC) (European Commission 1991).

Low cost sludge disposal methods no longer exist within the European Union. 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) (European Council 1999) there has been a sharp decline in landfilling of sludge, and in some EU states such as Germany (LeBlanc, Matthews et al. 2009) the practice has been banned completely unless the sludge is in the form of ash resulting from sludge incineration. There are also concerns over continued application of sludge to farmland. Agriculture farmers in countries such as Sweden have decided to stop the practice completely (Hultman, Levlin et al. 2000).

The result of an increase sludge volume and more stringent sludge disposal regulations means that the cost of sludge disposal has seen a substantial increase. Values in the literature for sludge transport and disposal costs vary between €100 and €200 per ton of dry solids (DS) depending on the final destination of the sludge (agriculture, composting, incineration) (Goldenman, Middleton 2008), and can account for between 15 and 20% of the total operational cost.

METHODOLOGY

A decision support tool (DST) was developed on the *Microsoft Excel VBA* platform. Nine treatment systems were included in the program (Table 3). Models were developed for each system. Each system model calculates labour costs, energy use, sludge production, chemical use, plant-footprint, OPEX, CAPEX, and LCC.

Capital expenditure

Due to the lack of site-specific capital expenditure data, values for CAPEX are limited to variations in scale only. Power law models were developed from data compiled and normalised to an Irish context (E.q.1) (Foess, Steinbrecher et al. 1998, Gkika, et al. 2014).

$$\mathrm{C}_c = \left(\frac{I_c C_t K_l}{I_t}\right) \times ER_l \qquad (1)$$

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 factor 2015 = 1.3), ER_l is the currency exchange rate (€ - US€, 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.

Gratziou et al. (Gratziou, et al. 2006) suggested a plant lifetime of 40 years. However, calculating life cycle costs of a range of plants over a 40-year period could be considered excessive. Most engineers currently plan for a design life of 25 years. Beyond this time, factors such as population and industrial growth, changes to water quality legislation, and environmental concerns are difficult to predict with reasonable accuracy. Thus, the design lifetime used here is 25 years.

Table 3. Nine treatment systems were used in the study.

Suspended growth	*Attached growth*	*Hybrid*	*Natural*
Complete mix activated sludge (CMAS)	Rotating biological contactors (RBC)	Integrated fixed-film activated sludge	Constructed Wetlands
Anoxic oxic (AO)	Trickling filters (TF)	Moving bed biofilm reactor (MBBR)	
Anaerobic anoxic oxic (AAO) - Sequence batch reactor (SBR)			

Energy

Energy modelling is limited to 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 (Metcalf & Eddy). Transport energy is not included here.

Aeration and pumping have been identified as the two main energy sinks that are influenced by changes in scale, load, discharge limits and temperature; and as such, were parameterised to reflect changes in conditions. The assumptions made for aeration and pumping models are presented below in Table 4.

Table 4. Modelling assumptions used for suspended growth systems

Parameter	*Value*
Average temperature	10°C (Ireland national average)
Average height above sea level	118 m (Ireland national average)
Alpha correction factor (α)	0.5 for BOD_5 removal only 0.65 for nitrification
Beta value (β) for DO saturation to clean water	0.95
Fouling factor (F)	0.9
Oxygen transfer rate (kg O_2/kWh)	Fine bubble diffusers – 3.5 Course bubble diffusers – 1.5
Pump efficiencies	0.75
Motor efficiencies	0.9
Cost of electricity (€ /kWh)	0.2

Labour

Labour-hours calculations are based on the methodology proposed by the *New England Interstate Water Pollution Control Commission* (NEIWPCC 2008). The labour categories and description are presented below in Table 5. The salary values given below reflect the average current salaries for wastewater operators, engineers, lab workers and yard hands/helpers in Ireland.

Table 5. Labour categorisation, description and cost.

Labour type	*Description*	*Cost per hour (€/hour)*
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

Sludge

Sludge production values are presented as kg DS/day. Table 6 gives the sludge solids concentrations assumed for the study. It is assumed that the solids concentration of fine-screen sludge from extended aeration systems is similar to that of primary treatment. Mechanical based treatment systems are assumed to employ mechanical sludge thickening and dewatering.

Constructed wetlands are assumed to have sufficient surface area availability for sludge drying beds. It should be noted that the values calculated for sludge disposal refer only to the cost of sludge disposal by contractor. Other costs related to sludge are included in chemical, labour and energy costs. The cost of contractor sludge disposal in this study is estimated to be €20 / m^3 sludge.

Table 6. Sludge dry solids concentrations assumed for the study.

Sludge type	*Dry solids (DS) concentration (%)*
Primary	6
SBR	5
Waste activated	0.8
Dewatered	18
Drying bed	24

Life cycle cost

The life cycle costing is calculated using the net present value (NPV) method (E.q.2). The interest rate assumed is 3.5%.

$$NPV = \text{InitialCost} + \sum_{k=1}^{N} FutureCost_k \left[\frac{1}{(1+i)^{n_k}}\right] \tag{2}$$

Where the initial cost is the capital investment in year 0, n is the year of expenditure, k is the item of expenditure and $i$is the interest in the year n.

The hydraulic definition of 200 L/PE is used in this study. In total, 15 tests were run to study the effect of variations in scale, loading and discharge limits. Plant design capacity was varied from 500 to 2000 PE. Variations in loading varied from high, to medium, to low strength wastewater as per the definition given by (Henze 2008). Variations in discharge limits are presented below in Table 7.

Table 7. Discharge limits applied.

	High DL	*Med DL*	*Low DL*
BOD	30	30	15
COD	100	100	80
TSS	35	35	20
TN	N/A	20	10
TP	N/A	5	2
NH3	N/A	5	0.5
PO43	N/A	2	0.5

DISCUSSION

As mentioned previously, the DST provides values for energy, chemicals, sludge production, and plant footprint; however, for the purpose of demonstration the discussion will be limited to OPEX and NPV. All systems exhibited economies of scale with respect to (w.r.t.) CAPEX, OPEX, and NPV. CAPEX outputs were differentiated only by scale, variations in organic loading and discharge limits were not reflected in the CAPEX component of the NPV, and therefore, the magnitude of change in the NPV w.r.t. variations in organic loading and discharge limit can only be attributed to changes in OPEX. Figure 1 shows the percentage change in NPV from high (560 mg

BOD_5/L), to low (230 mg BOD_5/L) organic loading for the three different plant sizes. It can be seen that as design capacity is reduced the effect of changes in loading on NPV become less significant. The IFAS and CW systems are the least affected by changes in loading. This could indicate that these systems may be suited for plants that experience large variations in organic loading. However, the IFAS system has one of the highest NVPs (Figure 2), and although the CW system has the lowest NPV, if surface area is restricted it may eliminate CW from consideration.

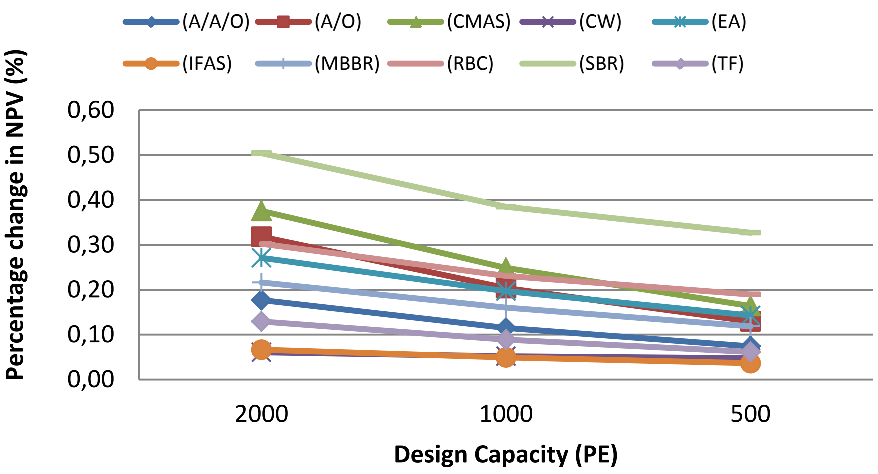

Figure 1. Percentage change in NPV from high to low organic loading with medium level discharge limits.

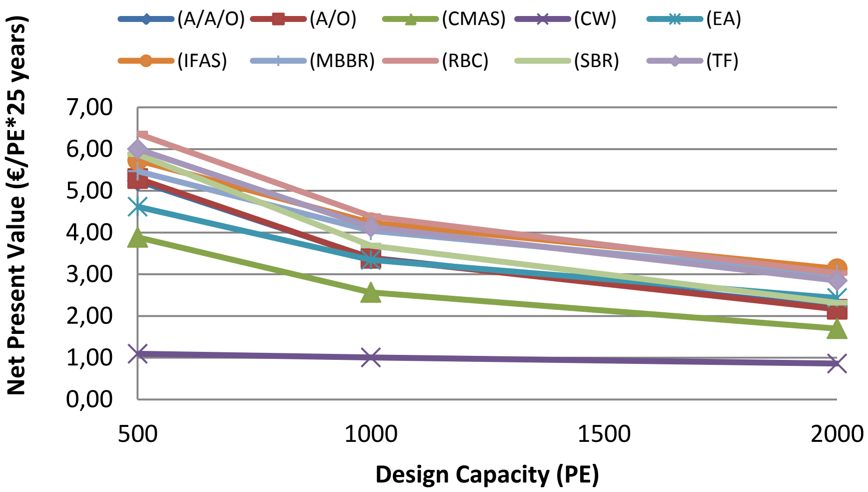

Figure 2. Net present value for high loading and average discharge limits.

Operational expenditure

Operational expenditure is dominated by the cost of labour for all systems. Because labour is a function of scale and system only, there is no change in OPEX w.r.t load or discharge limits. Labour costs increased with an increase in systems components at a fixed plant scale. This means hybrid systems, with more unit processes and unit components had much higher labour-hour requirements. Although the specific cost of labour did not change with variations in load and limits, the percentage contribution of labour cost to the overall OPEX was reduced with increase in load and lower limits due to increases in sludge quantities, energy and chemical demand (Figure 3).

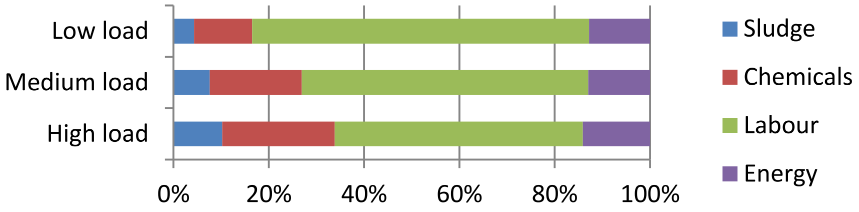

Figure 3. Effect of variation in load on the percentage contribution of OPEX components for MBBR systems

The rate of change in cost (€/PE-year) (Figure 4), was greater between 500 and 1000 PE, than between 1000 and 2000 PE. Constructed wetlands have the lowest OPEX per capita and the largest percentage increase with reduction in scale, but variations in loading had a negligible effect on operational cost. Systems with higher energy expenditure such as EA and IFAS exhibited large percentage increases in OPEX as the design capacity was reduced. Higher percentage increases in OPEX were observed at lower organic loads. A full breakdown of OPEX variation with organic load and scale is presented below (Table 8).

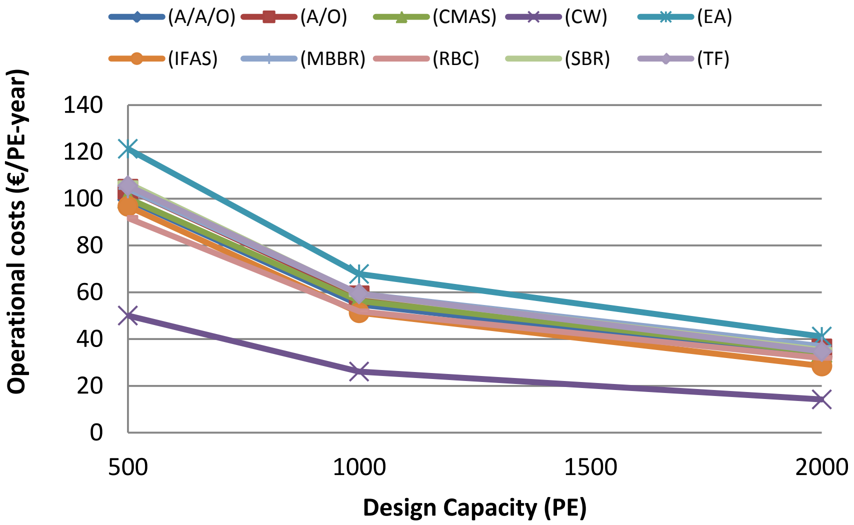

Figure 4. Operational expenditure (€/PE-year) with variation in scale with medium organic and medium level discharge limits.

Table 8. Operational expenditure (€/PE-year) with variations in scale (2000 - 500 PE) and organic load.

	High Load				*Medium Load*				*Low Load*			
PE	2000	1000	500	Δ %	2000	1000	500	Δ %	2000	1000	500	Δ%
(A/A/O)	35	57	102	191	33	55	99	206	30	53	97	220
(A/O)	40	63	109	169	36	58	104	191	32	55	101	212
(CMAS)	38	60	104	171	34	56	100	192	31	53	97	213
(CW)	15	26	50	246	14	26	50	252	14	26	50	257
(EA)	46	72	126	176	41	68	121	195	38	64	118	213
(IFAS)	29	52	98	233	29	51	97	239	27	50	95	254
(MBBR)	41	63	108	165	37	59	104	182	33	56	100	203
(RBC)	38	59	100	164	32	52	92	188	27	47	86	216
(SBR)	43	69	120	178	35	59	107	202	29	52	97	232
(TF)	37	61	108	192	35	59	105	204	32	57	103	218

CONCLUSION

- The objective of this study was to develop a decision support tool to assist with decision making during the procurement process. The use of a decision support tool that allows scenario-specific parameterisation of wastewater treatment systems can provide a better understanding of the potential for cost reduction. This approach becomes more beneficial with a decrease in plant capacity where economies of scale and variations in load become much more significant.
- The specific cost values used in this study (sludge disposal, chemicals, labour, and electricity) will vary by location. These cost components have been parameterised in the support tool in order to facilitate location-specific variation. However, it would be prudent to carry out sensitivity analyses to assess the impact of any future variation in cost.
- Further work is required to reduce some of the assumptions made in the study. Capital expenditure values need to be adjusted to reflect changes in discharge limits and organic loading. Aeration energy used in activated sludge based systems can account for up to 75% of total energy used by a plant, which places a greater weight of importance on assumptions of oxygen transfer efficiencies.

FURTHER WORK

In addition to work required improve the level of detail in the support tool; future work will also include the addition of a life cycle assessment component to evaluate the environmental impact associated with choice of system.

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