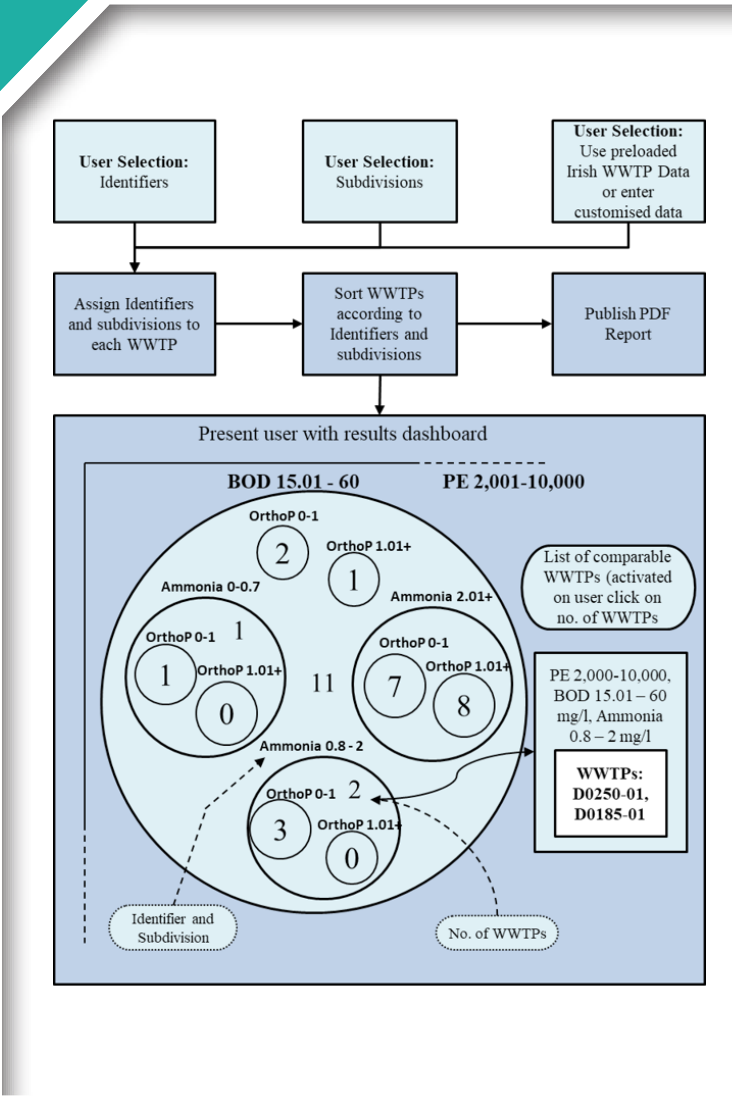
Report No. 255

epa Research
Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Optimal Design and Operation of Small-scale Wastewater Treatment Plants: The Irish Case

Authors: Lorna Fitzsimons, Greg McNamara, Edelle Doherty and Eoghan Clifford



www.epa.ie

Rialtas na hÉireann
Government of Ireland

ENVIRONMENTAL PROTECTION AGENCY

The Environmental Protection Agency (EPA) is responsible for protecting and improving the environment as a valuable asset for the people of Ireland. We are committed to protecting people and the environment from the harmful effects of radiation and pollution.

The work of the EPA can be divided into three main areas:

Regulation: *We implement effective regulation and environmental compliance systems to deliver good environmental outcomes and target those who don't comply.*

Knowledge: *We provide high quality, targeted and timely environmental data, information and assessment to inform decision making at all levels.*

Advocacy: *We work with others to advocate for a clean, productive and well protected environment and for sustainable environmental behaviour.*

Our Responsibilities

Licensing

We regulate the following activities so that they do not endanger human health or harm the environment:

- waste facilities *(e.g. landfills, incinerators, waste transfer stations)*;
- large scale industrial activities (*e.g. pharmaceutical, cement manufacturing, power plants)*;
- intensive agriculture *(e.g. pigs, poultry)*;
- the contained use and controlled release of Genetically Modified Organisms *(GMOs)*;
- sources of ionising radiation *(e.g. x-ray and radiotherapy equipment, industrial sources)*;
- large petrol storage facilities;
- waste water discharges;
- dumping at sea activities.

National Environmental Enforcement

- Conducting an annual programme of audits and inspections of EPA licensed facilities.
- Overseeing local authorities' environmental protection responsibilities.
- Supervising the supply of drinking water by public water suppliers.
- Working with local authorities and other agencies to tackle environmental crime by co-ordinating a national enforcement network, targeting offenders and overseeing remediation.
- Enforcing Regulations such as Waste Electrical and Electronic Equipment (WEEE), Restriction of Hazardous Substances (RoHS) and substances that deplete the ozone layer.
- Prosecuting those who flout environmental law and damage the environment.

Water Management

- Monitoring and reporting on the quality of rivers, lakes, transitional and coastal waters of Ireland and groundwaters; measuring water levels and river flows.
- National coordination and oversight of the Water Framework Directive.
- Monitoring and reporting on Bathing Water Quality.

Monitoring, Analysing and Reporting on the Environment

- Monitoring air quality and implementing the EU Clean Air for Europe (CAFÉ) Directive.
- Independent reporting to inform decision making by national and local government *(e.g. periodic reporting on the State of Ireland's Environment and Indicator Reports)*.

Regulating Ireland's Greenhouse Gas Emissions

- Preparing Ireland's greenhouse gas inventories and projections.
- Implementing the Emissions Trading Directive, for over 100 of the largest producers of carbon dioxide in Ireland.

Environmental Research and Development

- Funding environmental research to identify pressures, inform policy and provide solutions in the areas of climate, water and sustainability.

Strategic Environmental Assessment

- Assessing the impact of proposed plans and programmes on the Irish environment *(e.g. major development plans)*.

Radiological Protection

- Monitoring radiation levels, assessing exposure of people in Ireland to ionising radiation.
- Assisting in developing national plans for emergencies arising from nuclear accidents.
- Monitoring developments abroad relating to nuclear installations and radiological safety.
- Providing, or overseeing the provision of, specialist radiation protection services.

Guidance, Accessible Information and Education

- Providing advice and guidance to industry and the public on environmental and radiological protection topics.
- Providing timely and easily accessible environmental information to encourage public participation in environmental decision-making *(e.g. My Local Environment, Radon Maps)*.
- Advising Government on matters relating to radiological safety and emergency response.
- Developing a National Hazardous Waste Management Plan to prevent and manage hazardous waste.

Awareness Raising and Behavioural Change

- Generating greater environmental awareness and influencing positive behavioural change by supporting businesses, communities and householders to become more resource efficient.
- Promoting radon testing in homes and workplaces and encouraging remediation where necessary.

Management and structure of the EPA

The EPA is managed by a full time Board, consisting of a Director General and five Directors. The work is carried out across five Offices:

- Office of Environmental Sustainability
- Office of Environmental Enforcement
- Office of Evidence and Assessment
- Office of Radiation Protection and Environmental Monitoring
- Office of Communications and Corporate Services

The EPA is assisted by an Advisory Committee of twelve members who meet regularly to discuss issues of concern and provide advice to the Board.

EPA RESEARCH PROGRAMME 2014–2020

Optimal Design and Operation of Small-scale Wastewater Treatment Plants: The Irish Case

(2014-W-DS-16)

EPA Research Report

Prepared for the Environmental Protection Agency

by

Dublin City University

Authors:

Lorna Fitzsimons, Greg McNamara, Edelle Doherty and Eoghan Clifford

ENVIRONMENTAL PROTECTION AGENCY
An Ghníomhaireacht um Chaomhnú Comhshaoil
PO Box 3000, Johnstown Castle, Co. Wexford, Ireland

Telephone: +353 53 916 0600 Fax: +353 53 916 0699
Email: info@epa.ie Website: www.epa.ie

©Environmental Protection Agency 2018

ACKNOWLEDGEMENTS

This report is published as part of the EPA Research Programme 2014-2020. The EPA Research Programme is a Government of Ireland initiative funded by the Department of Communications, Climate Action and Environment. It is administered by the Environmental Protection Agency, which has the statutory function of co-ordinating and promoting environmental research.

The authors would like to acknowledge the EPA for its financial support. The authors acknowledge the contribution of the project steering committee, namely Mr John Gray (Consultant), Mr Brendan Kissane (EPA), Dr Eadaoin Joyce (Irish Water), Mr Robert Kennedy (Irish Water) and Dr Hadyn Love-Knowles (ex Irish Water).

DISCLAIMER

Although every effort has been made to ensure the accuracy of the material contained in this publication, complete accuracy cannot be guaranteed. The Environmental Protection Agency, the authors and the steering committee members do not accept any responsibility whatsoever for loss or damage occasioned, or claimed to have been occasioned, in part or in full, as a consequence of any person acting, or refraining from acting, as a result of a matter contained in this publication. All or part of this publication may be reproduced without further permission, provided the source is acknowledged.

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

EPA RESEARCH PROGRAMME 2014–2020
Published by the Environmental Protection Agency, Ireland

ISBN: 978-1-84095-785-3 August 2018

Price: Free Online version

Project Partners

Dr Lorna Fitzsimons
Mechanical and Manufacturing Engineering
Dublin City University
Dublin
Ireland
Tel: +353 1 700 7716
Email: lorna.fitzsimons@dcu.ie

Greg McNamara
Mechanical and Manufacturing Engineering
Dublin City University
Dublin
Ireland
Tel: +353 1 700 7716
Email: greg.mcnamara5@mail.dcu.ie

Dr Eoghan Clifford
Civil Engineering
College of Engineering and Informatics
NUI Galway
Galway
Ireland
Tel: +353 9 149 2219
Email: eoghan.clifford@nuigalway.ie

Edelle Doherty
Civil Engineering
College of Engineering and Informatics
NUI Galway
Galway
Ireland
Tel: +353 9 149 2219
Email: e.doherty4@nuigalway.ie

Contents

List of Figures

List of Tables

List of Boxes

Executive Summary

Background

Wastewater treatment systems have significant economic and environmental benefits associated with their construction and operation; however, like other such infrastructure, the economic and environmental impacts of their construction and operation need to be minimised. Economic costs, as well as capital, operational, and environmental impacts, can vary with location because of specific site conditions. Variations in scale and loading and discharge limits can all affect the performance of a treatment system and solutions to optimise performance can often be site specific.

One key challenge is selecting the most appropriate treatment system for a given location. This requires an understanding of how each of the competing systems will perform in a given scenario and how variation in performance influences each of the associated costs. Small agglomerations in particular face unique challenges in relation to their wastewater treatment needs. Internationally, it has been acknowledged that small-scale wastewater treatment plants can be resource intensive when compared with larger plants. Many of the scale economies associated with wastewater treatment significantly increase as agglomeration scales fall below 2000 population equivalent (PE). Historically, the main focus in choosing wastewater treatment systems has been to reduce capital costs while ensuring that the required effluent standards are met. However, because of the need to minimise the operational costs associated with wastewater treatment plants and their resource consumption, while meeting regulations, there has been a growing focus on their life cycle cost and environmental performance. It is, therefore, evident that appropriate economic and environmental assessment tools that consider the life cycle cost of wastewater treatment plants holistically may assist in the system selection process.

The key objectives of this research were to develop software tools to assist in the selection and management of wastewater treatment systems, with a specific focus on small wastewater treatment plants. The developed research software tools, the decision support tool (DST) and KPICalc respectively, provide (1) a framework and toolkit for assessing the life cycle costs of several wastewater treatment systems (economically and environmentally) and (2) a benchmarking toolkit to facilitate wastewater treatment plant management.

Summary of Key Findings and Outcomes

- Two software tools, DST and KPICalc, were developed.
- Effective and well-designed sampling of influent and effluent is key to ensuring that benchmarking is both accurate and useful. This report found that grab sampling may not provide data of sufficient quality for system operation and management.
- Based on the specific user inputs, and from an economic life cycle perspective, constructed wetlands were found to be a viable alternative to conventional electro-mechanical systems in locations where land availability at a reasonable cost is not an issue. In addition, constructed wetlands have a more favourable environmental profile due to reduced energy and chemical requirements.
- Variations in plant scale, organic and inorganic loading, and discharge limits have a significant effect on the economic and environmental performance of electro-mechanical treatment systems.
- Economic and environmental economies of scale were evident for most electro-mechanical systems. Scale economies are not as significant in constructed wetland systems because of the fixed linear relationship between population equivalent and required unit area.
- Energy use contributes significantly to both the operational cost and environmental impact. The magnitude of the environmental impact is as much a function of the electrical grid mix as of the quantity of energy used.
- There are significant trade-offs between regional and global environmental impact categories. This is largely dictated by a plant's discharge limits, i.e. more stringent discharge limits will reduce regional impact (eutrophication, toxicity) while increasing global impact (global warming, acidification).

Summary of Key Recommendations

- Economic cost evaluations and comparisons should be carried out over the system lifetime using appropriate life cycle costing models.
- Broader environmental impacts can be considered within design criteria, particularly in situations where the life cycle costs of competing systems are within a margin of uncertainty (it is assumed chosen designs will be suitable to meet discharge limits).
- Sludge disposal costs are location-specific and alternative disposal options should be assessed during life cycle cost assessment for competing systems.
- Constructed wetlands should be afforded due consideration in locations where land availability is not a limiting factor, subject to supporting business case analysis of competing processes on a case-by-case basis.
- Benchmarking of wastewater treatment plants should be further developed as a performance optimisation and management tool. This will require, in particular, the use of flow-proportional sampling at wastewater treatment plants.
- The development of methodologies to determine minimal data requirements for effectively implementing performance management, life cycle analysis and benchmarking tools should be considered.

1 Introduction

The water and energy nexus is rightly receiving attention from policymakers and researchers in an effort to promote joined-up thinking on sustainable energy and water supplies. The energy sector uses significant amounts of water to source and convert primary energy supplies, and the water and wastewater treatment sectors use significant amounts of energy to deliver and treat water and wastewater. From the global perspective, 2–3% of the world's energy is used for water supply and sanitation purposes (Olsson, 2012).

Energy requirements for wastewater treatment are a function of several variables, including scale, influent wastewater flow rates and concentrations, effluent water discharge requirements, technology selection, operating practices and control strategies. However, it is important to note that energy is not the only significant cost of operating wastewater treatment plants (WWTPs); other important cost categories include labour, chemicals and sludge management. When selecting treatment technology options, the cost of ownership, comprising both capital costs and operational costs, should be assessed and optimised across all cost categories and over the entire lifetime of the WWTP. Furthermore, several of these categories contribute to the broader environmental impact of WWTPs and technology selection should be considered carefully to mitigate adverse environmental effects.

The focus of this project is small WWTPs, defined for the purposes of this report as WWTPs with a population equivalent (PE) < 2000. Ireland has a particular landscape of WWTPs, with the vast majority of Irish plants being classified as small. For example, there are more than 500 agglomerations in Ireland between 50 and 500 PE, and all of these are subject to the Waste Water Discharge Authorisation Regulations. Recent research reported that approximately 87% of Irish WWTPs treat agglomerations of less than 10,000 PE (Shannon *et al*., 2014). The wide distribution of small WWTPs in Ireland reflects the low population densities beyond the larger cities and towns. Internationally, decentralised WWTPs are generally unstaffed (i.e. they may not have operational/maintenance staff based on site) and can be more likely to be infrequently maintained and have limited monitoring and reporting capabilities (O'Reilly *et al*., 2011). Choosing the optimal design for new plants and operating existing plants effectively and efficiently are complex matters and depend on a number of factors and variables. A non-exhaustive list includes scale; technology; loadings; discharge limits; WWTP location; discharge location; local climate, flooding propensity and topology; variation in incoming flows (industrial, agricultural, domestic, mixed, etc.); management priorities; feasibility of retrofitting economically; land availability; sludge management; monitoring capability; operational expertise; and cost.

The aim of this research, which builds on previous research funded by the Environmental Protection Agency (EPA) (Fitzsimons *et* al., 2016), was to develop methodologies, models and tools to assess, compare and benchmark the performance of small WWTPs from a number of perspectives: life cycle cost, environmental performance, resource consumption (energy and chemicals) and sludge management. This study also investigated how wastewater sampling can be optimised to provide improved data for life cycle assessment (LCA), benchmarking and other design/operational tools. The overarching aim should be to operate plants that treat wastewater in accordance with designated standards at an acceptable environmental and economic cost. However, it is not easy to benchmark, compare or predict individual plant performances, considering that WWTPs vary in scale, use different technologies or technology configurations, treat to achieve different effluent standards and accept influents with differing compositions and concentrations.

1.1 Objectives

The key objectives of this project were to:

1. conduct and collate detailed research into the state-of-the-art of sustainable, holistic, low-energy, solutions for the wastewater treatment industry, including the specific technologies that address the challenges of small indigenous WWTPs;

2. further optimise a key performance indicator (KPI) benchmarking toolkit (KPICalc) initially proposed in a previous project (Fitzsimons *et al*., 2016);
3. develop a methodology and framework to holistically assess the life cycle costs and environmental impact of representative small-scale WWTPs;
4. develop a decision support tool (DST) for life cycle costing and assessing environmental impact.

1.2 Project Outputs

The key outputs of this project include tools and models that can support policymakers, regulators, plant operators and researchers to manage existing WWTPs and to assess and compare the life cycle costs and environmental performance of wastewater treatment systems (WWTSs). Specifically, these outputs include:

- KPI benchmarking software tools (KPICalc);
- life cycle cost and life cycle environmental models of representative Irish WWTPs;
- a DST.

1.3 Report Structure

The report is broken down into a number of chapters and sections. Chapter 2 presents an overview of the relevant literature. Chapter 3 details the methodology and model development. Chapter 4 presents and discusses the key results of the research. Finally, the project conclusions and recommendations are presented in Chapter 5 and Chapter 6, respectively.

2 Literature Summary

2.1 Introduction

The International Water Association (IWA) specialist group on small WWTPs has defined small plants as those serving agglomeration sizes of below 2000 PE or processing influent flow rates of below 200 m^3/day (Lens *et al.*, 2001). The population distribution in Ireland is such that 587 WWTPs serve agglomerations of below 2000 PE.[1] Small agglomerations face particular challenges in relation to their wastewater treatment needs. Typically, at these agglomeration sizes, the per capita cost of wastewater treatment begins to show an exponential increase with decreasing scale. WWTSs have economic and environmental costs associated with their construction and operation. These costs vary with location because of the specific conditions under which a treatment plant must be built and operated. Variations in scale, organic loading, discharge limits, topography, temperature and other regional factors can influence system performance to the extent that the suitability of a particular system for a given location may be less than that of a competing system. Qualitative design criteria, such as robustness and reliability, will eventually be reflected in a system's total life cycle cost (LCC). Therefore, if the optimum system design can be considered as that which achieves the desired final effluent quality for the lowest cost, the choice of system requires an understanding of how alternative systems will perform under certain conditions and how variation in performance will ultimately influence cost. Conventional WWTSs can achieve high levels of pollutant removal; however, the performance of each system will vary depending on the type and quantity of substrate to be removed, and thus the appropriate solution is always contextual and situational (Schumacher, 1989). This suggests there is no "one-size-fits-all" solution applicable to every location or for every location where there is a system, or a system configuration that will outperform others. The main problem is how to determine which system is most appropriate for a particular location. It has been suggested that the selection of the most appropriate wastewater treatment technology is the biggest challenge faced by wastewater treatment management (Molinos-Senante *et al.*, 2015).

2.2 Wastewater Treatment Systems and Processes

Conventional WWTSs generally fall into one of four categories: suspended growth, attached growth, hybrid and natural systems. Table 2.1 presents just some of the systems commonly found in operation.

2.2.1 Natural wastewater treatment systems

Natural WWTSs are low energy consumers that require large surface areas over which to operate. They require minimal human or material input and

Table 2.1. Categories of wastewater treatment systems and processes

Suspended growth	Attached growth	Hybrid	Natural
Conventional activated sludge (CAS)	Rotating biological contactors (RBCs)	Membrane bioreactor (MBR)	Constructed wetlands (CW)
Anoxic oxic (AO)	Trickling filter (TF)	Moving bed biofilm reactor (MBBR)	Reed bed
Anaerobic anoxic oxic (AAO)	Membrane aerated biofilm reactor	Integrated fixed-film activated sludge (IFAS)	Waste stabilisation pond
Sequence batch reactor (SBR)	Pumped flow biofilm reactor	CAS/TF	Aerated lagoon
Extended aeration (EA) – oxidation ditch (OD)	Horizontal flow biofilm reactors	RBC/reed bed	Soil filters, sand filters

1 EPA-supplied data.

thus can be suited to isolated locations (Solano *et al*., 2004; Babatunde *et al*., 2008; Kayranli *et al*., 2010). The percentage of operational expenditure (OPEX) attributable to labour has been reported to be higher for small WWTPs (Table 2.2) (Reicherter, 2003). McEntee (2006) reported that CW maintenance costs can be as little as 1/20th of those for a conventional system and, therefore, these systems may prove to be more economical than conventional systems in certain situations.

Although often referred to as low-tech systems, the mechanism by which pollutant removal is carried out in natural systems is complex and specialised. Each natural treatment system type has specific strengths and limitations that make them better suited to particular locations and conditions. The choice of system will depend largely on the required effluent quality and land availability. Vertical flow (VF) CWs have a greater capacity to remove ammonia with a reported required surface area of 3 m^2/PE, while horizontal flow (HF) CWs have a greater capacity for total nitrogen (TN) removal but require a much greater surface area (10 m^2/PE) (Wallace and Knight, 2006). Phosphorus removal in CW systems has had limited success and an additional mechanical process may be required where low-phosphorus effluent is required. Combinations of natural systems are often integrated to produce a particular effluent quality by utilising pollutant removal mechanisms specific to each system type (Belmont *et al*., 2004).

2.2.2 Conventional electro-mechanical systems

Treatment is normally carried out through a series of unit processes that remove solids and pollutants through filtration, gravity settling, biological decomposition or chemical precipitation. Each unit process requires material and energy inputs, the quantities of which vary depending on system type, system configuration and site-specific conditions. Suspended growth systems employ energy-intensive aeration to achieve biological oxidation of the wastewater substrate. The level of aeration required will depend on a plant’s organic loading and discharge limits. Ammonium removal may require up to 30% more aeration energy and TN reduction will require varying levels of material and energy depending on system configuration.

Table 2.2. Percentage of OPEX attributable to labour for a range of plant sizes (Reicherter, 2003)

PE	Percentage of OPEX attributable to labour
< 10,000	35–40
10,000–100,000	25
> 100,000	15

Data sourced and adapted from Reicherter (2003).

While most suspended growth systems utilise chemicals to achieve phosphorus removal, biological phosphorus removal can be achieved with the AAO system; however, when very low phosphorus effluent concentrations are required, combinations of biological and/or chemical processes, as well as other processes (e.g. absorption), may be required. Attached growth systems achieve biological aeration through different means and, therefore, have different energy sinks. RBCs deliver the biomass to atmospheric air on partially submerged discs that rotate continuously in and out of the wastewater. The motors that drive disc rotation are the main energy sink in RBC systems. TFs distribute the wastewater over an attached growth media filter bed and may require a significant pumping energy input. A typical TF energy consumption value of 0.093 kWh/m^3 has been reported (Metcalf & Eddy, 2014); however, this value is likely to increase as system size decreases. Furthermore, depending on the desired loading rates, a TF system may require forced draft aeration, which introduces another energy sink to TF systems. As with suspended growth systems, nutrient removal with attached growth systems requires additional energy and material inputs. Phosphorus removal using attached growth systems has had limited success and generally requires chemical input. Hybrid WWTSs are designed to achieve a specific final effluent quality by combining processes from different systems. The IFAS system combines the robustness of attached growth systems with the flexibility of suspended growth systems to achieve high-quality effluent in cases where available surface area is minimal (Johnson *et al*., 2006).

In general, conventional electro-mechanical WWTSs can be more material and energy intensive than some natural systems (von Sperling, 1996; Gratziou *et al*., 2006; McEntee, 2006; Kalbar *et al*., 2013; Rawal and

Duggal, 2016), although, as mentioned above, this will depend on site-specific requirements.

2.3 Life Cycle Cost Analysis

The term “life cycle cost” (LCC) was first introduced in 1965 in a report entitled “Life cycle costing in equipment procurement” (LMI, 1965). The report determined that the cost of system acquisition may be low in relation to the cost of system ownership (Eisenberger and Lorden, 1977). Early cost assessment methods used for system selection often centred on the initial capital expenditure (CAPEX) (Woodward, 1997), with secondary consideration given to operation and maintenance costs. This approach was limited and did not always recommend the most cost-effective system. WWTPs in particular have widely varying, system-specific operational costs, which, when considered over the lifetime of the system, can outweigh any differences in initial CAPEX. Life cycle cost analysis (LCCA) is a holistic approach used to assess the economic feasibility of a system over the entirety of its predicted lifetime (Arditi and Messiha, 1999). The LCCA 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 (Dhillon, 2009). The net present value (NPV) method is commonly used to calculate LCCs (equation 2.1).

$$NPV = Initialcost + \sum_{k=1} FutureCost_k \left[\frac{1}{(1+d)^{n_k}} \right] \quad (2.1)$$

where the initial cost is CAPEX in year 0, n is the year of expenditure, k is the item of expenditure and,d is the discount rate. Several studies have demonstrated the value of the LCCA approach in WWTS cost assessment. Rawal and Duggal (2016) found that LCCA was most suitable for solving complicated WWTS selection issues such as resource allocation and maintenance management. Johnson *et al*. (2006) employed LCCA to elucidate many of the cost factors that must be considered when considering IFAS system implementation, such as land, growth media and construction cost indices. Pretel *et al*. (2016) combined LCCA and LCA to illustrate the economic and environmental performances that can be achieved with anaerobic MBRs. The recurring theme in these studies was the significant variation in OPEX from system to system and the variation in the CAPEX/OPEX ratio. These studies underlined the importance of considering both CAPEX and OPEX over the system’s lifetime.

2.4 Life Cycle Assessment

LCA is an analytical tool that provides a holistic approach to assessing the environmental performance of a product or system from cradle to grave (Tillman and Baumann, 2004). In Ireland the LCA approach has been widely accepted as a valid environmental assessment tool by government, academia and parts of the private sector (Table A1.2 in Appendix 1 presents details of some of the LCA studies conducted in Ireland to date). The EPA has stated that “In the coming decade, businesses will increasingly be required through regulatory approaches to undertake life cycle assessment for their goods and services and to adopt eco-label standards” (EPA, 2016). Public agencies are compiling data to assist with this effort. In Ireland, a Carbon Management Tool has been developed to help businesses assess their carbon output (Irish Environment, 2016). In Northern Ireland, the Waste Management Strategy is based on an LCA that considers how waste can be minimised and recovered at every stage in production processes (Irish Environment, 2016). The application of LCAs to wastewater treatment is particularly appropriate owing to the nature of the relationship between a plant’s technosphere and the surrounding ecosphere. The application of an LCA to a WWTP was first reported in The Netherlands in 1997 to examine the sustainability of municipal wastewater treatment (Roeleveld *et al*., 1997). The study concluded that improvements in the environmental performance of wastewater treatment should focus on minimising effluent discharge pollutants and sludge production, and that the impact from energy consumption was negligible. However, Gallego *et al*. (2008) subsequently concluded that the environmental loading associated with energy production for use in the treatment process was one of the main contributors to the overall environmental profile of WWTPs. Since the initial study in The Netherlands, over 40 LCA studies had been published in peer-reviewed journals up to 2013 (Corominas *et al*., 2013). These studies covered a variety of objectives, which included assessing changes in system configuration (Tillman *et al*., 1998), variations in boundaries and scale (Lundin *et al*., 2000), structural changes (Vidal *et al*., 2002) and competing technologies (Kalbar *et al*., 2013). In recent times, there has been a paradigm shift in environmental assessment of treatment systems from considering not only water quality and human health but also energy and resource recovery (Corominas *et al*., 2013).

2.5 Wastewater Treatment Plant Management and Performance Assessment

Across Europe, challenges surrounding management practices and the education and training of staff in the water sector have been exacerbated by issues regarding ageing infrastructure and a lack of efficient wastewater treatment (Heino *et al*., 2011). These challenges are also present to varying degrees within the Irish context; population is on the rise and there is an ongoing need to upgrade WWTPs that do not meet regulatory requirements. Furthermore, there may be a need to further upgrade WWTPs to meet the potential future demand to remove pathogens and emerging contaminants (e.g. pharmaceutical products). Ensuring regulatory compliance while operating in an environmentally sustainable manner (among other responsibilities) can result in WWTP management becoming a complex process. Performance assessment and benchmarking have developed as a key aspect of WWTP management and, when systematically applied, these tools can be effective in the management of WWTPs.

Benchmarking is the systematic process of searching for best practices and effective operating procedures that lead to increased performance and the subsequent adaptation of these practices to improve the performance of an organisation (Parena and Smeets, 2001; Cabrera *et al*., 2009). Benchmarking is a data-driven process and can be successful only if careful consideration is given to data availability and accuracy. Improved data management practices can be achieved through WWTP performance assessment and benchmarking; WWTPs in countries that have employed benchmarking for several years have expanded their data acquisition and reliability. To address data availability during the early years of a performance assessment or benchmarking project, it is necessary to identify interim methods and acquisition strategies to maximise the potential initially and thereafter the ongoing benefits of benchmarking. The data utilised in this study were taken from public sources (e.g. annual environmental reports) and measured by the research team at various sites.

Two important areas affect the ability of managers to efficiently and effectively operate WWTPs: (1) process monitoring and (2) challenges associated with data acquisition, storage and analysis. Performance assessment methodologies provide solutions to these issues while also facilitating WWTP benchmarking and performance improvement. Typical performance assessment and performance improvement (benchmarking) processes are shown in Figure 2.1. Repetition of these processes will promote continuous improvement of WWTP performance.

Continual performance assessment and benchmarking can be implemented in WWTPs using software applications, the benefits of which include reduced WWTP manager/operator workload and process standardisation. However, WWTP benchmarking can be complicated; accurate and reliable data are essential (Lindtner *et al*., 2008) and benchmarks require careful interpretation (Torregrossa *et al*., 2016).

Flow monitoring is one of the key data sources for WWTPs, and is provided for in most larger and more recently constructed WWTPs in Ireland, The operation and maintenance of flow meters is required by regulation [under the Environmental Protection Agency Act (1992)] and issues with flow monitoring (e.g. meters that are improperly installed, outside calibration or inadequately maintained) are reported in annual environmental reports (e.g. EPA, 2015). In 2015, EPA audits identified 68 WWTPs with no continuous flow meter in place on the effluent stream and 51 WWTPs that failed to provide a composite sampler for compliance monitoring purposes (EPA, 2015).[2] Historically, poor calibration and a lack of provision of flow meters were identified as general issues in Irish WWTPs, which resulted in many WWTPs being unable to collect accurate flow data. Inaccurate or unavailable flow metering is a key impediment to benchmarking, as it is a fundamental parameter for many standard KPI metrics. It should be noted that a capital programme and a business plan to address these issues was

2 https://www.water.ie/news/proposed-capital-investme/Proposed-Capital-Investment-Plan-2014-2016.pdf

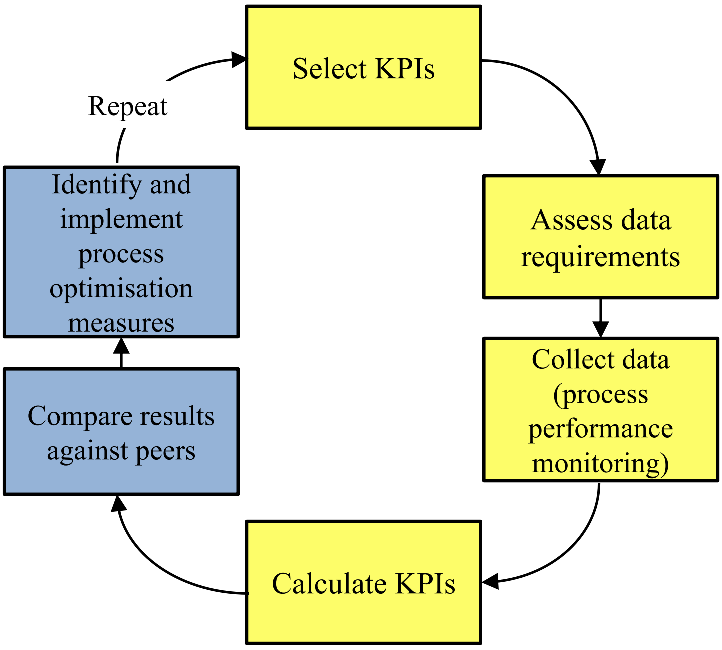


Figure 2.1. Typical performance assessment (yellow) and performance improvement/benchmarking (blue) processes using key performance indicators (KPIs).

put in place by Irish Water.[3] Following asset survey reports, WWTPs in need of an upgrade are being identified and improvement works carried out. The use of mobile sampling units in some areas can also be considered if appropriate.

Energy consumption in WWTPs is influenced by a number of factors, including WWTP size (economies of scale) (Mizuta and Shimada, 2010), influent characteristics and discharge limits (which define the degree of pollutant removal required to meet compliance) and the technologies employed (Yang *et al.*, 2010) along with the age and condition of the asset/technologies, and historic and current operation and maintenance regimes. Accounting for all of these influences when reporting energy consumption as part of WWTP performance benchmarking can be challenging (Longo *et al.*, 2016).

Large-scale WWTPs are reported to have some economic benefits over small-scale WWTPs, including, in some cases, a reduction in energy consumption per unit of wastewater treated (Mizuta and Shimada, 2010). The cost of wastewater treatment per PE per year has been reported to decrease with increasing WWTP design capacity in WWTPs in Switzerland (Thaler, 2009). A study of the energy consumption in WWTPs in China reported that energy consumption per unit of wastewater treated decreased with increasing WWTP loading due to the scale effect during the operation of WWTPs (Tao and Chengwen, 2012). In addition, energy consumption results [kWh/m^3 treated and kWh/kg chemical oxygen demand (COD) removed] were affected by the percentage of design capacity utilised in a WWTP (Tao and Chengwen 2012). Because of the scale effect, comparing energy consumption KPI results across WWTPs of varying scale is not a valid approach. While the reported energy consumption results often consider the volume of wastewater treated at each WWTP, the kWh/m^3 metric cannot take into account the effects of scale or the extent to which design capacity is being utilised.

Many WWTP performance benchmarking methods require wastewater pollutant concentration data, which may often be scarce (Rieger *et al.*, 2010; Talebizadeh *et al.*, 2016). This scarcity is linked to the high cost, in terms of workload and financial resources, associated with long-term experimental collection of wastewater quality data (Martin and Vanrolleghem, 2014). Flow-paced sampling typically provides the most representative means of collecting wastewater data; however, its application in many small-scale WWTPs

3 https://www.water.ie/docs/Irish-Water-Business-Plan.pdf

may not be feasible because of the high costs and maintenance associated with flow meter and sampler management. In some WWTPs, such as those that employ sequencing batch reactors (SBRs) as part of the treatment process, influent is delivered in regular batches, which can result in reduced influent variability. Therefore, some WWTPs may be able to employ more cost-effective sampling methods, such as grab sampling, to collect representative samples.

Carlson and Walburger (2007) observed that comparing energy use between WWTPs can be a valuable exercise for performance improvement schemes when peer or comparable WWTPs are correctly identified by their load and operational conditions. This requirement applies across all WWTP KPIs, as benchmarking between WWTPs should occur only if the WWTPs are comparable (Matos *et al*., 2003). Many factors (e.g. WWTP size, discharge regulations) can complicate the identification of comparable WWTPs for benchmarking purposes.

2.6 Conclusion

This chapter has considered and reviewed some of the approaches associated with determining optimal system selection for small WWTPs from both an economic and an environmental perspective. In addition, the challenges associated with comparing and benchmarking WWTP performance, such as variability in scale and technology, data availability and data accuracy, have been outlined.

3 Methodology and Toolkit Development

3.1 Economic and Environmental Life Cycle Cost Analysis

An overview of the procedure adopted for this study is presented in Figure 3.1. It should be noted that the primary objective of this section of the report is to demonstrate a framework/methodology that can be used to assist with the WWTS selection process, specifically by quantifying the main economic and environmental cost components in a life cycle context. Where possible, specific cost data have been acquired from Irish sources, but many of the data are taken from international sources. However, the specific cost data could be considered arbitrary, as they are required only to assist with the demonstration of the methodology.

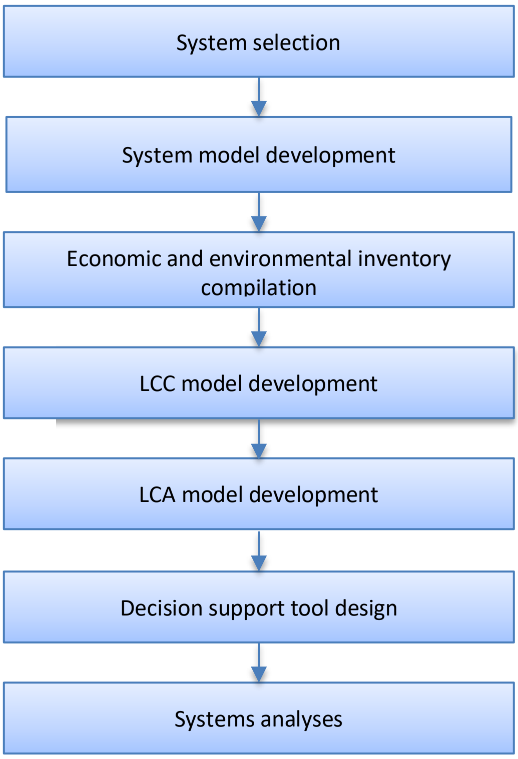


Figure 3.1. Overview of study procedure.

3.1.1 System selection

In order to identify suitable WWTSs for inclusion in the study, a survey was conducted of the 538 licensed WWTPs in Ireland (see Table A1.1). Suspended growth systems are the most common system type in Ireland, accounting for almost 60% of all systems. Of these, CAS systems account for over 36%, with EA,[4] SBR, IFAS and MBBR systems accounting for the remainder. Attached growth systems (excluding hybrid IFAS and MBBR systems) account for less than 10%. Biofilter, pump flow bioreactors and MBR systems collectively account for just over 1%. Integrated constructed wetlands (ICWs) are the primary natural system type and account for less than 0.5% of all systems. Reed beds are commonly used as a tertiary treatment stage in some smaller plants but could not be considered for inclusion as a stand-alone system. Several factors were considered in relation to the systems that should be included in the study. The selected systems should be representative of Irish WWTSs or those that might feasibly be used in small-scale applications. The quantity of data required for the economic life cycle cost inventory (LCCI) and the environmental life cycle inventory (LCI) was also considered. If sufficient system-specific data could not be sourced, the system was excluded.

An ICW design is generally bespoke and heavily dependent on regional geography and topography, which makes it challenging to present any standard design format for these systems. Furthermore, ICW systems provide additional benefits beyond that of a WWTS, such as the ability to treat agricultural run-off, restoring potentially lost environmental infrastructure, and providing public amenities and tourist attractions. These additional benefits are difficult to capture within an economic and environmental cost assessment model. Nevertheless, it was determined that a natural system should be included to illustrate the cost savings that can be achieved when land availability is not a limiting factor and land cost is not excessive. Therefore, a hybrid horizontal flow–vertical flow

4 The percentage of EA systems also includes oxidation ditches. The terms were used interchangeably throughout the survey of plants.

(HF-VF) CW system with nitrification and denitrification capacity was included in the study. The IFAS and MBBR systems can be easily retrofitted to existing CAS, AO, and AAO systems and provide a means to increase capacity without significantly increasing surface area requirements. However, both systems are quite similar in construct and operation, with the main difference being the lack of a return activated sludge line in the MBBR system. It was determined that the inclusion of both would add little to the study and, therefore, only the IFAS system has been included. The systems included in the study are presented in Box 3.1, and general system schematics are presented in Figure 3.2 (note that each system can include a variety of internal recycle lines, step-feeds, external carbon sources or return activated sludge lines in various positions; for clarity, these are not included).

3.1.2 System model development

Reviews of the literature have found that, for most systems, over 90% of OPEX can be attributed to energy, labour (including laboratory work and administration), sludge disposal and chemicals (US EPA, 1981; Wendland, 2005). The cost of replacement parts for natural, suspended growth and attached growth systems, over their life cycles, was reported to range from 3% to 10% of the initial CAPEX (Rawal and Duggal, 2016). Therefore, the OPEX target quantities considered in this study were energy consumption, chemical use, sludge production and labour hours. System modelling of complete mix activated sludge (CMAS),[5] AO, AAO, EA, IFAS and SBR systems was based on the methodologies presented by Metcalf & Eddy (2014). The OD model is as presented by Davis (2010) and the TF and RBC system models are sourced from Metcalf & Eddy (2014). CW modelling was based on studies conducted by Gikas *et al*. (2014) and Vymazal (2005, 2010).

Box 3.1. Selected systems
Treatment system
Single stage complete mix activated sludge
Integrated fixed-film activated sludge
Anoxic oxic
Anaerobic anoxic oxic
Constructed wetland
Trickling filter
Rotating biological contactor
Extended aeration
Oxidation ditch
Sequence batch reactor

3.1.3 Economic data acquisition

CAPEX data for CMAS, AO, AAO, RBC, and SBR systems were sourced from Foess *et al*. (1998), for TF and OD systems from Gratziou *et al*. (2006), for IFAS systems from Johnson *et al*. (2004) and for CW systems from Gkika *et al*. (2014). Power law CAPEX cost curves were developed from the compiled data and normalised to reflect equivalent costs in Ireland. Operational cost data, such as the specific costs of chemicals, energy and sludge disposal, were compiled from several sources in Ireland.

3.1.4 Life cycle cost model

Figure 3.3 presents the LCC cost distribution framework for the systems' LCC models. Combinations of present value (PV) methods are used to estimate the LCCs. The single present value (SPV) method estimates one-off or infrequent costs such as parts replacement and end-of-life residual value. The uniform present value (UPV) method is used to estimate recurring operational costs. The individual PVs are combined with the initial CAPEX to yield the total LCC. Details of the LCC calculation methods and representative data values used in the model are provided in sections A1.2 and A1.3.

3.1.5 Life cycle assessment model

The LCA format adopted for the study adheres to the framework presented in the ISO 14040 series of standards (ISO, 1997, 1998, 2000; Lecouls, 1999) and references guidelines on the standards published by Guinée *et al*. (2001). The LCA software used is GaBi 6.0. The GaBi LCI database provided by Thinkstep

5 Conventional activated sludge (CAS), sometimes referred to as activated sludge (AS), is an umbrella term for all activated sludge processes. Complete mix activated sludge (CMAS) is a term used to differentiate it from the plug flow activated sludge process.

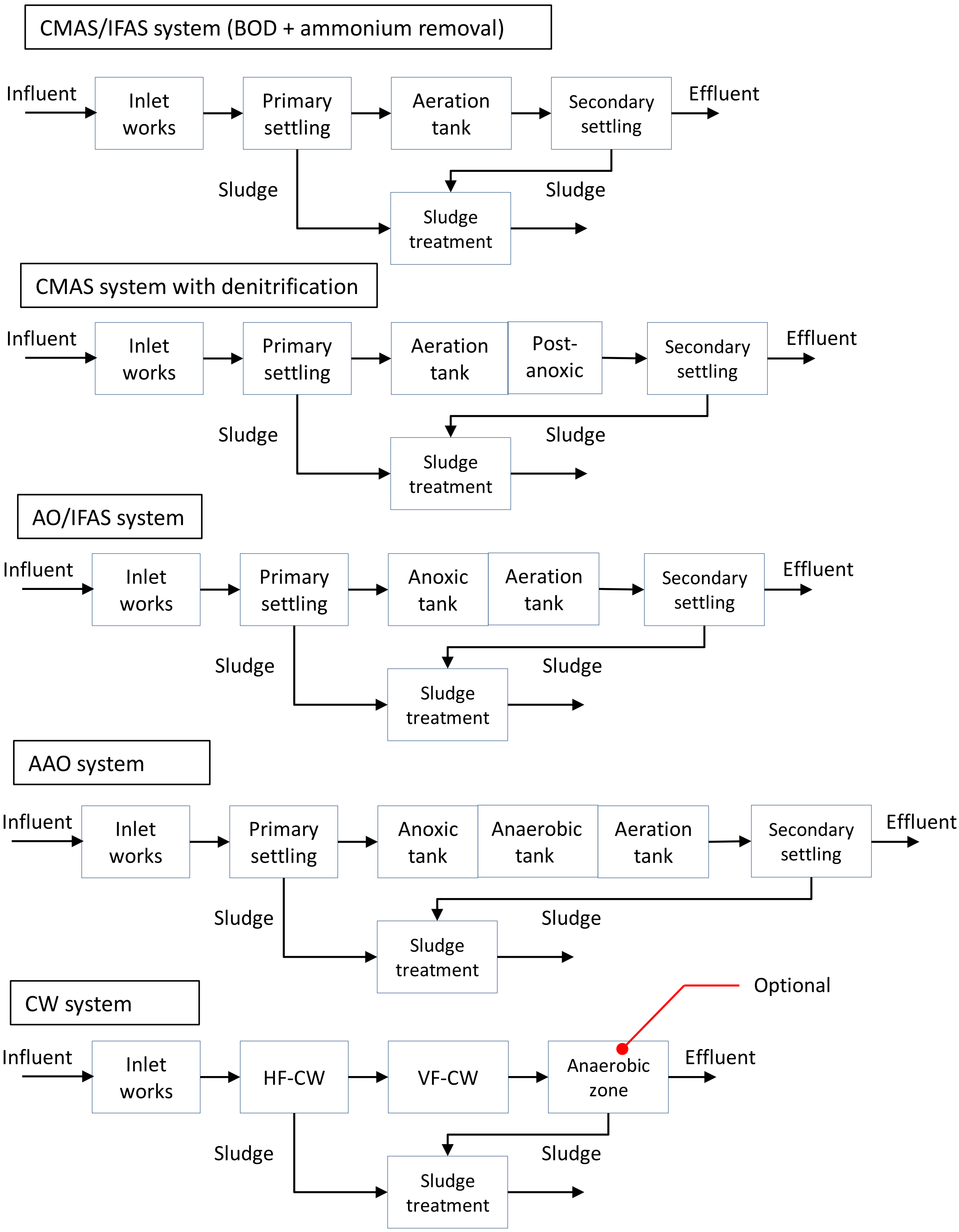


Figure 3.2. System schematics. BOD, biological oxygen demand.

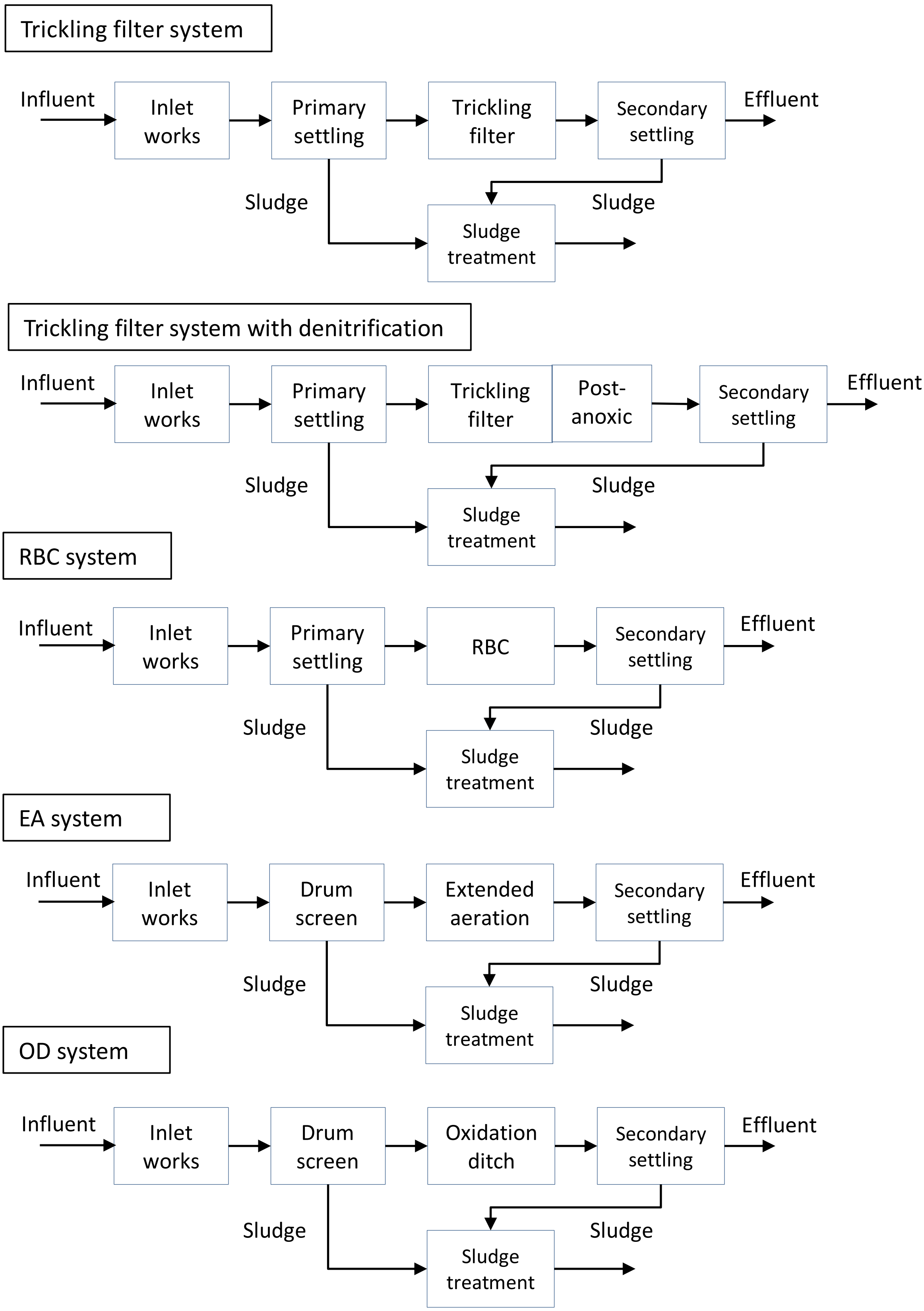


Figure 3.2. Continued.

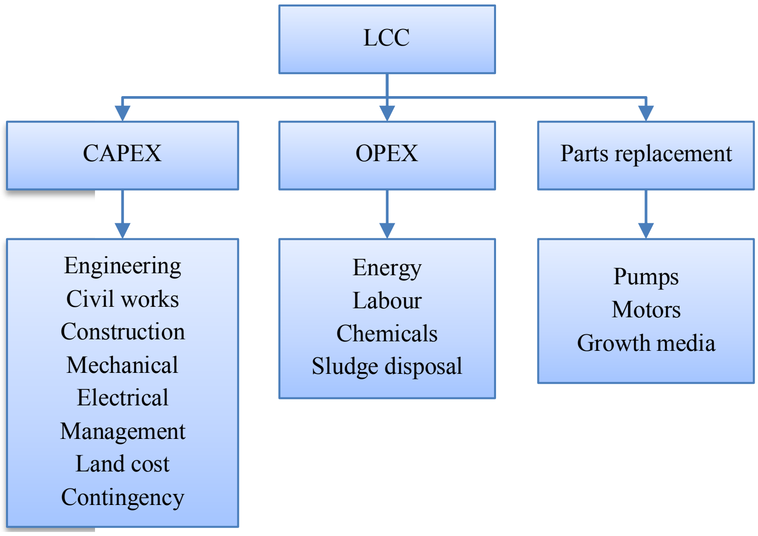


Figure 3.3. Life cycle cost distribution.

(formerly PE International) contains inventory data for upstream and downstream processes. The functional unit was defined as "1 day of operation". The boundaries of each system include all LCIs for energy and chemical production from cradle to grave, including sludge transport and application to land, and aerial and unit process emissions. The life cycle impact assessment methodology used in this study is that developed by the CML (Institute of Environmental Sciences, Leiden University) in 2001, which is compliant with the ISO 14040 series and has been adopted by authors of similar studies (Hospido *et al*., 2008). The impact category definitions for the CML methodology are presented in Table 3.1.

3.1.6 Decision support tool

The DST was developed on the Microsoft Excel 2010 platform. The program was designed to assist in WWTP selection process by providing economic, energetic and environmental system-specific information for a range of user-defined, site-specific scenarios. Previous research established that the parameters that have the largest influence on system performance and selection are scale, loading, discharge limits and land availability. These parameters have been soft coded to allow for user-defined, site-specific variability. In addition, the choice of sludge treatment and disposal was found to have a significant influence on operational cost and, therefore, it was determined that the toolkit should provide sludge treatment and disposal alternatives. An overview of the user input parameters and the program outputs is presented in Figure 3.4. The DST user interface is presented below (Figures 3.5 to 3.7).

Within the DST, the user inputs include plant loading, discharge limits and sludge treatment options. The program outputs life cycle CAPEX and OPEX costs, predicted energy and chemicals consumption values, predicted energy distributions, and environmental impact data, all on a system-specific basis. The tool also readily facilitates system comparison across a range of these parameters and over several metrics; for example, energy consumption can be compared with water treated (per cubic metre) or contaminant removed (per kilogram).

The program assesses WWTP steady-state performance and calculates all required operational quantities on a 24-hour-per-day basis. Energy modelling includes fine-bubble diffused aeration, horizontal surface aeration, TF pumping and RBC motor energy. Other pumping energy sinks, such as return activated sludge and nitrate recycling, have also been included. Energy sinks common to all systems, such as inlet works, primary and secondary sedimentation, volute operation and municipal energy (lighting, administration, building), have been estimated from the academic literature, engineering reports and manufacturers' design specifications. Chemical consumption and sludge production calculations are based on the user-defined discharge limits and loading. Labour estimates are based on data published by the New England Interstate Water

Table 3.1. CML 2001 life cycle impact assessment categories

Impact category	Abbreviation	Units
Global warming potential	GWP	kg CO_2 equiv.
Acidification potential	AP	kg SO_2 equiv.
Eutrophication potential	EP	kg PO_{43-} equiv.
Ozone depletion potential (steady state)	ODP	kg R11 equiv.[a]
Photochemical oxidation potential	PCOP	kg C_2H_6 equiv.
Ecotoxicity		kg $C_6H_4Cl_2$ equiv.
Freshwater aquatic toxicity potential	FAETP infinite	
Terrestrial toxicity potential	TETP infinite	
Marine aquatic toxicity potential	MAETP infinite	
Human toxicity potential	HTP infinite	kg $C_6H_4Cl_2$ equiv.
Abiotic depletion elements	ADPe	kg Sb equiv.
Abiotic depletion fossil	ADPf	MJ

[a]The refrigerant R11 is a chlorofluorocarbon (CFC).

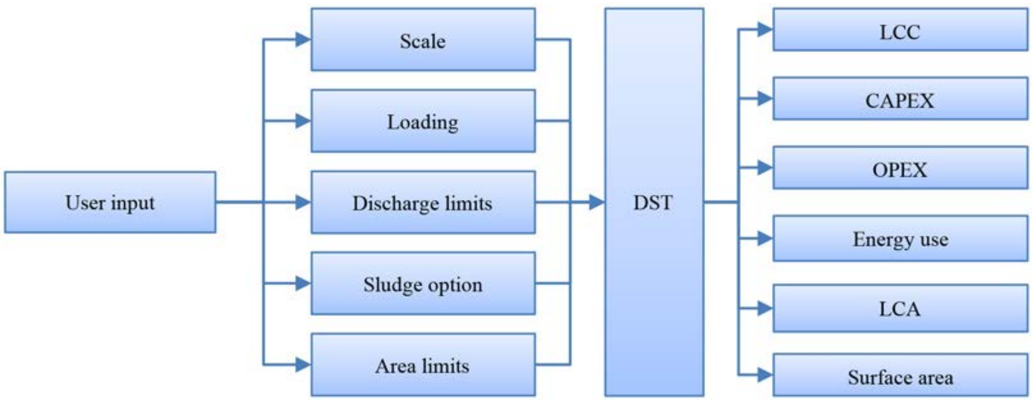


Figure 3.4. Decision support tool program overview.

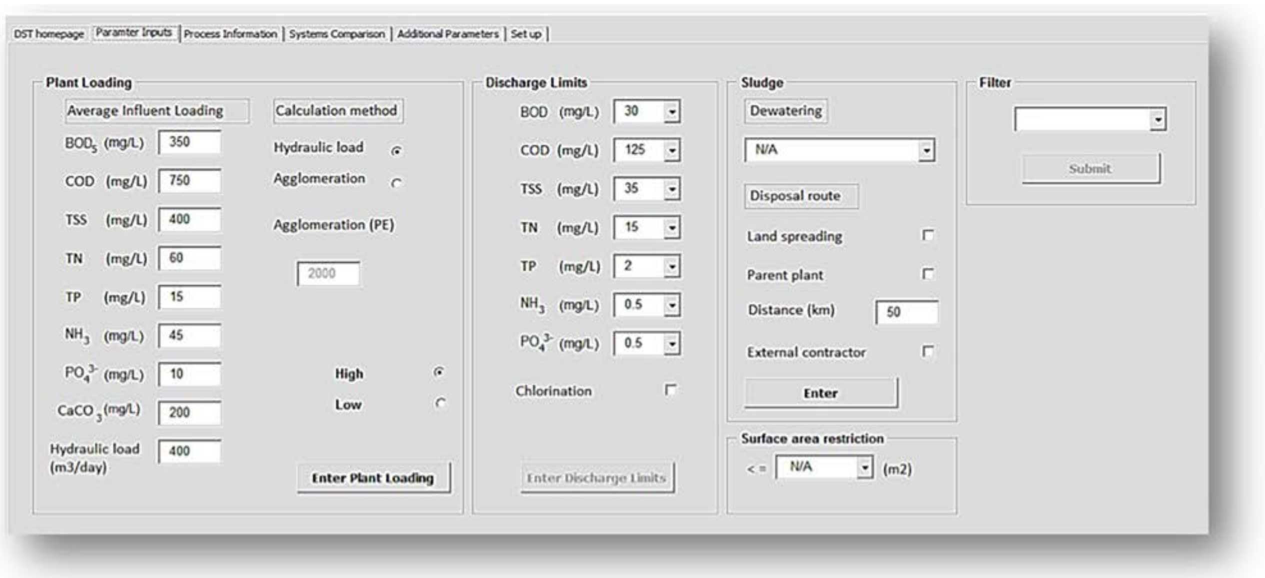


Figure 3.5. User input screen.

Figure 3.6. System information screen.

Figure 3.7. System comparison screen.

Pollution Control Commission (NEIWPCC, 2008). The program includes default values for specific operational costs, such as energy and chemicals, but also additional controls to allow for regional variance. LCCs are estimated for a 24-year system lifetime. Separate discount rates were used for energy PVs (5%) and OPEX PVs (3.5%) to allow for frequent variations in energy prices (these can be changed by the user). A system depreciation value of 3.5% was used to determine residual value at the end of the system lifetime. Additional quantities required for the environmental component include aerial emissions from unit processes and transport. Process emissions of CO_2, N_2O and CH_4 were based on the study conducted by Machado *et al*. (2007). Transport emissions from sludge disposal and chemical delivery were based on an average distance of 25 km but can also be user defined.

The results obtained from the DST presented and discussed in this report are based on specific user inputs, which are a combination of values from the literature and theoretical models. Relevant, detailed Irish data were requested from Irish Water, but these data were not available during the course of this research. However, the research undertaken and the developed model serve as a framework to assess and compare the effects of changes to key parameters for WWTSs in terms of OPEX and CAPEX, as well as environmental impact.

3.2 Performance Assessment and Benchmarking

From a review of existing literature, the need for a performance assessment methodology that can address the key gaps in the literature was clear. These gaps are summarised in Figure 3.8.

The most prominent challenge to successful performance assessment identified in the literature is that of data availability; without sufficient data, many of the other challenges in Table 3.2, such as assessing data accuracy and identifying comparable WWTPs, become increasingly complex, if not impossible. Each of the tools and methodologies developed in this research took account of data availability issues. The aim was to overcome and adapt to this challenge, in particular by allowing users to input their perception of how accurate data might be. In addition, the developed tools and methodologies can improve WWTP management practices when implemented at a national level.

To overcome the challenges identified in the literature, three methodologies and tools (Table 3.2) have been developed: (1) a performance assessment toolkit, KPICalc; (2) a comparable WWTP identification tool; and (3) a methodology for assessing the statistical agreement between wastewater sampling methods.

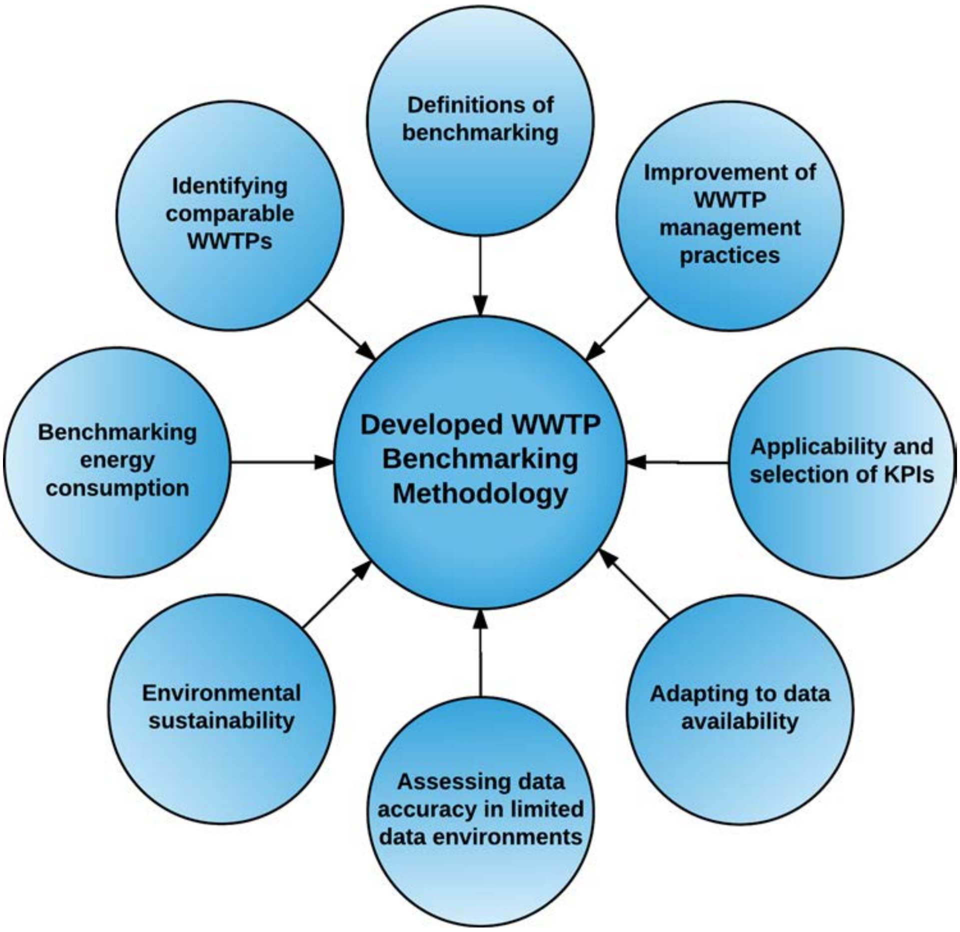


Figure 3.8. Challenges facing WWTP benchmarking methodologies.

Table 3.2. Developed methodologies and tools to overcome the challenges identified in the literature

	Definitions of benchmarking	Environmental sustainability	Adapting to data availability	Applicability and selection of KPIs	Assessing data accuracy in limited data environments	Benchmarking energy consumption	Improvement of WWTP management practices	Identifying comparable WWTPs
KPICalc performance assessment toolkit	✓	✓	✓	✓	✓	✓	✓	
Comparable WWTP identification tool			✓			✓	✓	✓
Sampling methods statistical agreement			✓				✓	

3.2.1 Performance assessment methodology development

The performance assessment methodology, KPICalc, was developed to address the challenges shown in Figure 3.8. KPICalc was developed using Microsoft Excel as the working platform because it has the data entry, calculation, graphing and coding capabilities required to implement and test the methodology and is widely used and available in the wastewater sector. KPICalc was designed to be used as a performance assessment tool or as a performance improvement tool when combined with the comparable WWTP identification tool. Figure 3.9 details the framework

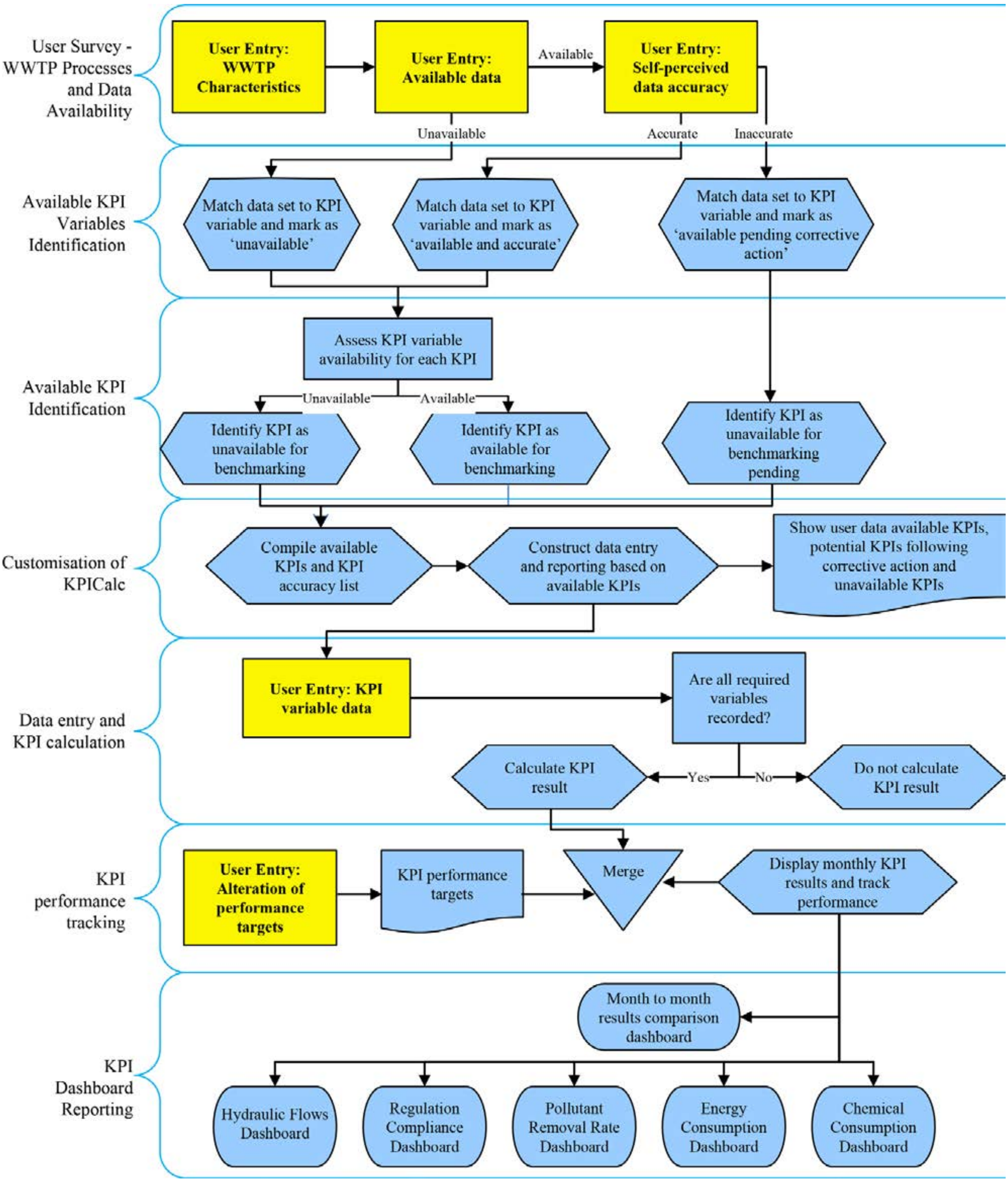


Figure 3.9. KPICalc framework.

behind KPICalc; modules that require user input are shown in yellow, with automated processes shown in blue.

Initially, the end user (engineer, facility manager, etc.) completes a short survey, which enables the automatic selection of KPIs to be used for the remainder of the KPICalc process. On completion of the survey regarding available data sources, the user is required to rate the accuracy of each data source. The user-perceived accuracy assessment step in the methodology provides an insight into the degree of data accuracy issues for a WWTP by highlighting the number of KPIs that cannot be included in benchmarking because of data accuracy issues. This incentivises the user to correct these data accuracy issues, which in itself can lead to improvements in WWTP performance.

Once the survey is complete, data sources are identified as available/unavailable and accurate/inaccurate and are matched to corresponding KPIs. Each KPI is then defined as available, unavailable or available pending corrective action. The customised (based on the survey) data entry module for WWTP data enables users to enter data as frequently as is desired or available. By default, KPICalc will calculate only KPIs based on data sources identified as accurate by the user. KPICalc utilises calculated KPI results to track the performance of the facility in relation to user-defined targets. KPICalc will also give headline information regarding the performance trends “performance improving”, “performance remaining steady” or “performance declining”, where each headline reports the change in performance from month to month. Reporting dashboards and PDF reports of KPI results are available to provide a means of focusing on results from one aspect of WWTP performance (e.g. energy usage).

3.2.2 Comparable wastewater treatment plant identification tool

Comparing the performance of WWTPs without first assessing the disparities between the operational conditions that might affect their performance can lead to incorrect deductions from performance assessments. Regular reassessment of WWTP comparability is required because of changing discharge licences and WWTP loadings and capacities; therefore, a Microsoft Excel-based tool was developed to identify comparable WWTPs in a rapid and standardised manner. Although this study focused on WWTPs with PE < 2000, this tool was designed to incorporate all WWTPs. A process diagram of the comparable WWTP identification tool is presented in Figure 3.10.

The identification of comparable WWTPs is based on WWTP design capacity (or WWTP loading) and the discharge limits with which a WWTP must comply. In the developed methodology, these WWTP characteristics are called identifiers. Each identifier is split into subdivisions to account for economies of scale (in the case of WWTP size primary identifiers) and the difficulty associated with meeting stringent discharge requirements (in the case of discharge limits). The toolkit recommends the use of carbonaceous biochemical oxygen demand (cBOD), ammonium and orthophosphate as the discharge limit identifiers, as these are most commonly applied discharge limits in WWTPs.

Initially, the user is asked to select comparable WWTP identifiers from a series of drop-down menus. The recommended identifiers are automatically selected; however, users can select alternative identifiers. The user must then select subdivisions for each identifier; again the recommended subdivisions are automatically entered and users can adapt these to suit their needs. Finally, the user can select the data preloaded in the tool for 355 licensed and operational Irish WWTPs, enter new data or make changes to the existing dataset.

Automated grouping of comparable WWTPs occurs instantly once the user has selected the identifiers, subdivisions and dataset. Based on the comparable WWTP identifiers and subdivisions selected by the user, the tool automatically assesses each WWTP in the dataset, assigns a series of identifiers and segments these results further based on the subdivisions selected. Following this, the tool creates groups (the number of groups equals the number of possible combinations of identifiers and subdivisions) and populates these groups with WWTPs that meet the characteristics of each group.

The comparable WWTP identification tool reports results to the user both in an interactive results dashboard format and as PDF reports. The results dashboard comprises a graphical representation (similar to a Venn diagram) of the results (Figure 3.10).

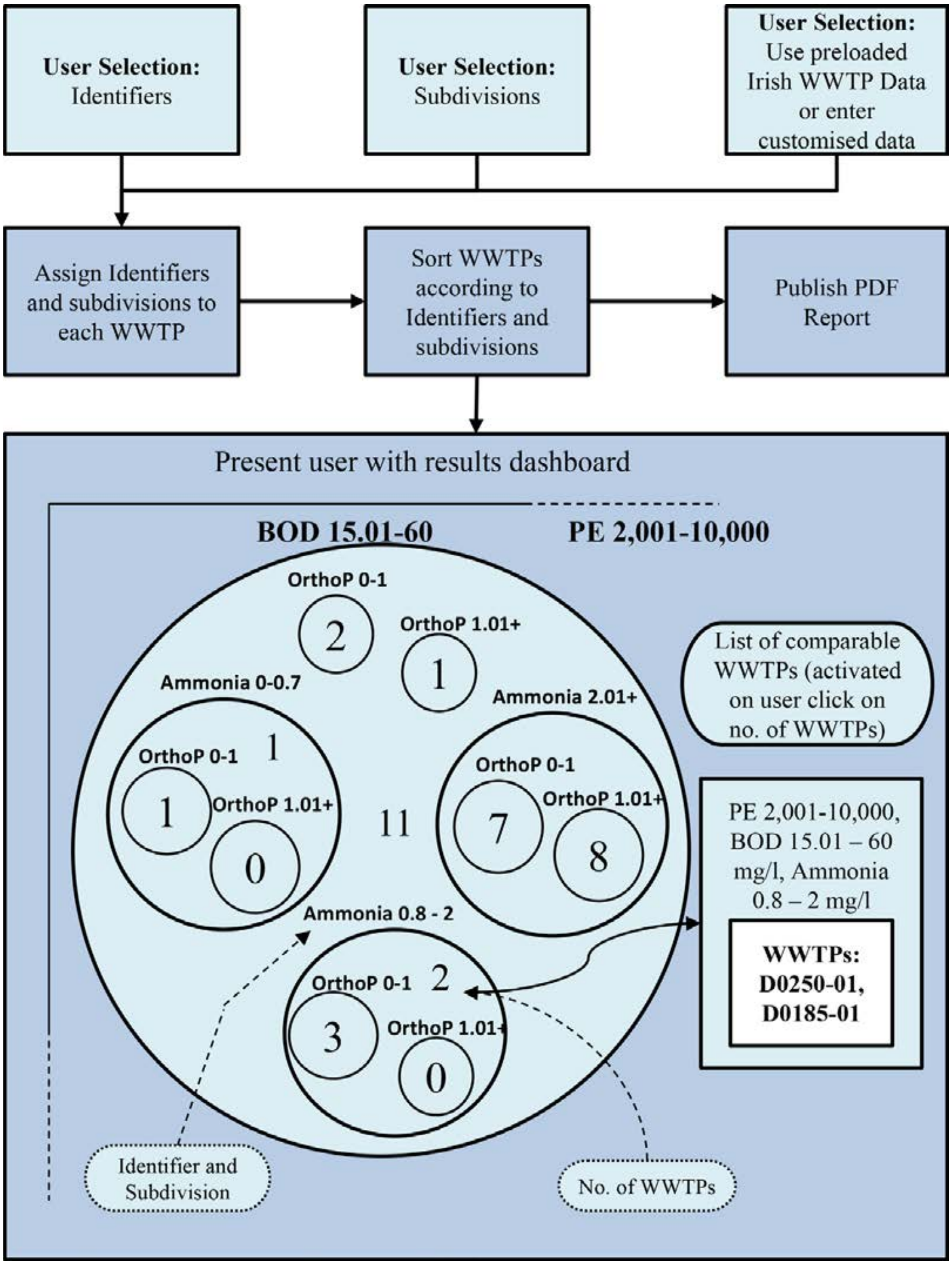


Figure 3.10. Comparable WWTP identifier tool process chart.

Each dashboard serves as a rapid method of assessing the number of WWTPs within a comparable group.

3.3 Assessment of the Statistical Agreement between Wastewater Sampling Methods for Daily WWTP Benchmarking Purposes

WWTPs commonly rely on the use of wastewater sampling and laboratory analysis methods to quantify the concentration of pollutants in wastewater. Because it is useful to utilise wastewater quality data as part of the KPI calculation, it is necessary to ensure that the time series of the wastewater quality data matches that of the other variables used in the KPI calculation. Furthermore, the widespread use of pollutant concentration data in WWTP performance benchmarking requires that the data should be collected in a reliable and representative manner. Although flow-paced sampling is typically the most representative means of collecting wastewater data, its application in many small-scale WWTPs may not be feasible because of the high costs and maintenance associated with flow meter and sampler management. As a result, it was necessary to assess if an alternative, more feasible, method may be suitable for collecting data for WWTP performance benchmarking purposes. To do this, the statistical agreement of the daily mean concentrations of influent wastewater characteristics, obtained in two WWTPs (Table 3.3) was assessed using a number of grab-sampling methods and a flow-paced method (Figure 3.11).

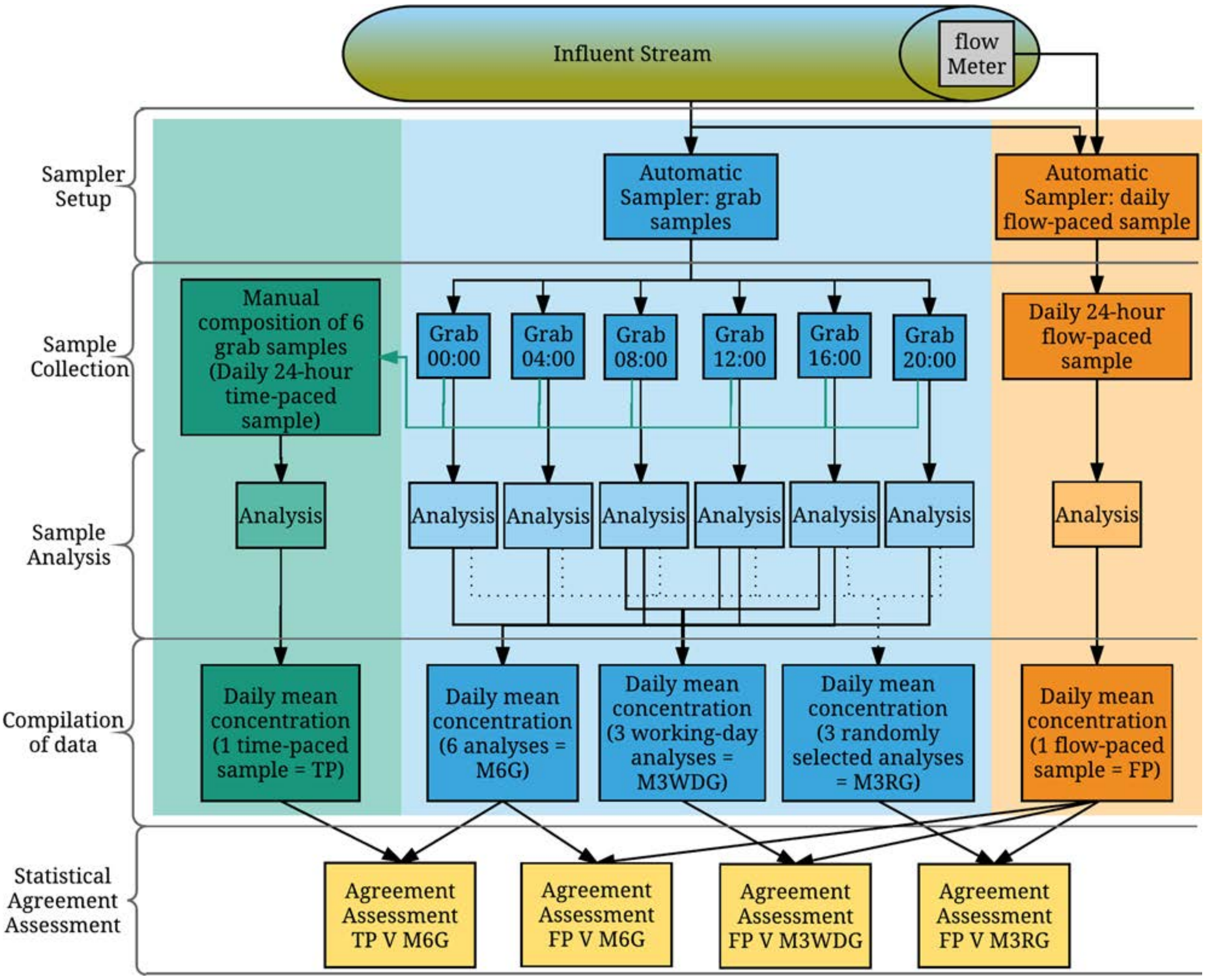


Figure 3.11. Flowchart of the daily sampling methods and agreement assessments of various methods.

Table 3.3. Wastewater treatment plant characteristics

Characteristic	WWTP A	WWTP B
Treatment technology	Activated sludge with phosphorus removal	Imhoff tank and TFs
Influent flow type	Batch delivery of influent from external pumping station to WWTP designed for continuous flow	Continuous flow
Influent characteristics	Municipal wastewater and imported sludge	Municipal wastewater only
Design capacity (as BOD)	24,834 PE	700 PE
Organic loading	13,640 PE	1483 PE
Hydraulic capacity (DWF) (m^3/year)	1,303,780	51,100
Hydraulic capacity (peak flow) (m^3/year)	1,847,995	153,300
Hydraulic loading (m^3/year)	1,959,788	162,522

BOD, biochemical oxygen demand; DWF, dry weather flow.

Table 3.4. Influent wastewater sampling details

Characteristic	WWTP A	WWTP B
Sampling dates	23/07/2016 to 24/08/2016	26/07/2016 to 24/08/2016
Number of days	30	30
Flow stream sampled	Influent	Influent
Automatic sampler (flow-paced)	ISCO 5800	ISCO 4700
Flow meter	ISCO 4250s	ISCO 4250s
Flow-paced sample details	One daily flow-paced composite sample (approximately 100 aliquots)	One daily flow-paced composite sample (approximately 100 aliquots)
Number of daily flow-paced samples collected	31	30
Automatic sampler (grab samples)	Aquacell S320	Aquacell S320
Grab sample details	6 grab samples (1000 ml) collected at 4-hour intervals	6 grab samples (1000 ml) collected at 4-hour intervals
Number of grab samples collected	180	179

Flow-paced composite sampling and grab sampling was carried out on the influent stream of two WWTPs of varying characteristics for approximately 30 days (Table 3.3).

Details of the sampling methodologies applied in the selected WWTPs are given in Table 3.4. Collected samples were analysed for COD, total suspended solids, ammonium-nitrogen and phosphate-phosphorus. The daily mean concentrations of the results from these sampling methods were assessed for agreement using Lin's concordance correlation coefficient (CCC).

4 Results and Discussion

4.1 Economic Life Cycle Cost Analysis and Environmental Life Cycle Assessment

The DST was developed and used to carry out economic LCCA and environmental LCA on 10 WWTSs with user input variations in scale, organic loading and discharge limits. The details of loading rates, discharge limit values and sludge disposal costs are provided in section A1.3. A subset of results is presented here to illustrate the variations in the economic and environmental performance of the WWTSs as a function of the user input. Agglomeration scales ranged from 500 to 2000 PE. Organic loading varied between high and low loading conditions, and discharge limits ranged from high (least stringent – BOD removal only)[6] to low (most stringent – BOD, NH_4, phosphorus and TN removal). Three sludge treatment and disposal options were included in the systems analyses:

- Option 1 involved sludge treatment with an all-in-one thickening and dewatering unit with polymer and lime addition. It was assumed that the sludge was removed from the WWTP site for land spreading.
- Option 2 involved sludge storage with no treatment and removal from site by an external contractor.
- Option 3 included sludge drying beds with lime addition for stabilisation and final removal by an external contractor.[7]

In all three sludge options, the final destination was assumed to be agricultural farmland, as this reflects the most common route in Ireland (Joyce and Carney, 2012).

It is important to stress at this point that many of the specific costs (e.g. chemicals and labour) and capital costs that have been used to demonstrate the methodology are based on national averages, international academic literature and personal communications. These costs can vary significantly with location and, therefore, in another location, a different set of region-specific costs could produce an entirely different set of results. Thus, the results presented here should be interpreted not as advocating one system over another but as a demonstration of how the LCCs of each of the considered systems are influenced by site-specific variation. The LCCs of all systems were calculated using the data in section A1.3. The LCCA determined that the CW system had the lowest LCC of all scenarios and, in many cases, was orders of magnitude lower than the LCC of electro-mechanical systems.[8] Therefore, the remainder of the LCC discussion focuses on the electro-mechanical systems. Tables 4.1 and 4.2 present the electro-mechanical systems with the lowest and highest LCC, respectively. The full set of results and analysis can be found in McNamara (2017). It should be reiterated that the LCC results presented here are based on several user-defined scenarios that were designed to demonstrate the effect of variation on site-specific conditions.

6 Biochemical oxygen demand is a surrogate measure of the mass of organic substrate in the wastewater. It is represented by the mass of oxygen required to oxidise the substrate. The term ‘BOD removal only’ is used to differentiate between systems that are required to remove BOD, nitrogen and phosphorus, and those that are required to remove BOD only.

7 Sludge drying beds are not very common in Ireland because they require significant surface areas, are prone to odour problems and generally perform better in more arid climates. However, in the correct conditions they can provide a low-cost alternative to mechanical sludge treatment.

8 It should be noted that a conservative land value of €5/m^2 was used in the study. In reality the cost of land could be as much as €50/m^2, as for the CW in Kill, County Waterford.

Table 4.1. Life cycle cost analyses (lowest LCC)

	Load	Sludge option 1			Sludge option 2			Sludge option 3		
		Scenario	System	LCC (€1 × 10^6)	Scenario	System	LCC (€1 × 10^6)	Scenario	System	LCC (€1 × 10^6)
DL band 4	High	S1	AAO	1.94	S25	TF	2.31	S49	AAO	1.85
	High	S2	AAO	2.83	S26	RBC	4.00	S50	AAO	2.66
	High	S3	AAO	3.58	S27	RBC	5.52	S51	AAO	3.33
	Low	S4	TF	1.73	S28	TF	1.87	S52	TF	1.67
	Low	S5	AAO	2.51	S29	TF	3.02	S53	AAO	2.38
	Low	S6	AAO	3.05	S30	TF	4.02	S54	AAO	2.87
DL band 3	High	S7	TF	1.89	S31	TF	2.20	S55	TF	1.80
	High	S8	AAO	2.92	S32	TF	3.87	S56	AAO	2.75
	High	S9	AAO	3.73	S33	TF	5.38	S57	AAO	3.48
	Low	S10	TF	1.67	S34	TF	1.81	S58	TF	1.60
	Low	S11	TF	2.40	S35	TF	2.90	S59	TF	2.28
	Low	S12	TF	2.99	S36	TF	3.84	S60	TF	2.80
DL band 2	High	S13	TF	1.78	S37	TF	1.99	S61	TF	1.71
	High	S14	TF	2.68	S38	TF	3.35	S62	TF	2.54
	High	S15	TF	3.42	S39	TF	4.57	S63	TF	3.21
	Low	S16	TF	1.62	S40	TF	1.71	S64	TF	1.56
	Low	S17	TF	2.30	S41	TF	2.66	S65	TF	2.18
	Low	S18	TF	2.81	S42	TF	3.47	S66	TF	2.66
DL band 1	High	S19	TF	1.74	S43	RBC	2.01	S67	TF	1.66
	High	S20	TF	2.49	S44	RBC	3.32	S68	TF	2.33
	High	S21	TF	3.10	S45	RBC	4.44	S69	TF	2.85
	Low	S22	TF	1.64	S46	TF	1.76	S70	TF	1.57
	Low	S23	TF	2.25	S47	TF	2.71	S71	TF	2.12
	Low	S24	TF	2.71	S48	TF	3.52	S72	TF	2.53

LCCs ranged from a low of €1.56 × 10^6 (TF system) to a high of €7.81 × 10^6 (EA system). Attached growth systems generally had the lowest LCCs in scenarios with less stringent discharge limits. The TF LCC values ranged from €1.56 × 10^6 to €5.38 × 10^6 as the specific input conditions varied. The AAO system showed the best performance of the suspended growth systems and had the lowest LCC in scenarios in which discharge limits were most stringent, regardless of organic load variation. The EA, OD and SBR systems had the highest LCCs. LCC values ranged from €2.28 × 10^6 to €7.81 × 10^6 for EA systems, from €2.22 × 10^6 to €7.33 × 10^6 for OD systems and from €1.77 × 10^6 to €3.31 × 10^6 for SBR systems. In scenarios without ammonia removal, LCC was lowest for TF systems. This was mainly due to the reduction in pumping energy requirements in scenarios involving removal of BOD only. Two contributing factors are considered here. Firstly, the specific organic loading rate (OLR) for BOD removal only, 0.6–2.4 kg BOD/m^3/day (Metcalf & Eddy, 2014), is much higher than for BOD and nitrogen removal (0.08–0.4 kg BOD/m^3 day), which means that a much greater volume of medium is required for nitrogen removal. This has the effect of increasing the distance the fluid has to travel by increasing static head height or extending distributor arm length. Secondly, a minimum hydraulic loading rate (HLR) is required to control wetting efficiency and biofilm thickness (0.5 l/m^2/s was adopted for this study) (Metcalf & Eddy, 2014). As the required volume of medium increases, the HLR decreases. As stated, nitrogen removal requires a greater volume of medium, and this generally results in HLRs below 0.5 l/m^2/s. To solve this problem, a portion of the TF effluent is recirculated and combined with the incoming primary effluent. The required recirculation ratio is measured relevant to the influent flow and ranges from 0.5 to 4.0 depending on the OLR (Metcalf & Eddy, 2014).

Table 4.2. Life cycle cost analyses (highest LCC)

	Load	Sludge option 1			Sludge option 2			Sludge option 3		
		Scenario	System	LCC (€1 × 10^6)	Scenario	System	LCC (€1 × 10^6)	Scenario	System	LCC (€1 × 10^6)
DL band 4	High	S1	SBR	2.20	S25	EA	2.95	S49	SBR	2.09
	High	S2	OD	3.36	S26	EA	5.46	S50	OD	3.19
	High	S3	OD	4.42	S27	EA	7.81	S51	OD	4.17
	Low	S4	SBR	2.00	S28	EA	2.40	S52	SBR	1.92
	Low	S5	SBR	2.75	S29	EA	4.10	S53	EA	2.67
	Low	S6	EA	3.49	S30	EA	5.63	S54	OD	3.26
DL band 3	High	S7	SBR	2.21	S31	EA	2.93	S55	SBR	2.11
	High	S8	SBR	3.31	S32	EA	5.45	S56	EA	3.13
	High	S9	OD	4.30	S33	EA	7.80	S57	OD	4.06
	Low	S10	SBR	1.98	S34	EA	2.38	S58	SBR	1.90
	Low	S11	SBR	2.74	S35	EA	4.07	S59	SBR	2.59
	Low	S12	OD	3.38	S36	EA	5.60	S60	OD	3.21
DL band 2	High	S13	SBR	2.10	S37	EA	2.72	S61	SBR	2.01
	High	S14	SBR	3.05	S38	EA	4.93	S62	EA	2.90
	High	S15	OD	3.90	S39	EA	6.99	S63	OD	3.70
	Low	S16	SBR	1.94	S40	EA	2.28	S64	SBR	1.86
	Low	S17	EA	2.67	S41	EA	3.84	S65	EA	2.56
	Low	S18	EA	3.30	S42	EA	5.23	S66	EA	3.14
DL band 1	High	S19	SBR	2.00	S43	OD	2.76	S67	SBR	1.90
	High	S20	OD	2.96	S44	OD	5.11	S68	OD	2.79
	High	S21	OD	3.78	S45	OD	7.33	S69	OD	3.54
	Low	S22	SBR	1.85	S46	OD	2.22	S70	SBR	1.77
	Low	S23	OD	2.50	S47	OD	3.77	S71	OD	2.44
	Low	S24	OD	3.16	S48	OD	5.16	S72	OD	2.98

Therefore, at very low OLRs it may be necessary to pump up to four times the average influent flow. The AAO system was the optimal choice in scenarios with phosphorus reduction requirements. There are several contributing factors. Firstly, there are reductions in phosphorus-precipitating chemical requirements as a result of enhanced biological phosphorus removal. Secondly, the inclusion of a pre-anoxic tank reduces the oxygen demand as oxygen is released during nitrate reduction, thus lowering aeration energy. Alkalinity is also restored during pre-anoxic denitrification, thus lowering the requirement for the addition of external alkalinity. Finally, the AAO system does not require the addition of external carbon. However, it should be noted that variation in flow rates is more significant in very small systems than in large systems. Controlling an AAO system in these conditions can be challenging, and close monitoring by experienced personnel is necessary.

4.1.1 Variation in scale

The effect of increasing scale is a reduction in the percentage of the systems' LCC that is attributed to CAPEX. Figure 4.1 presents the LCC distribution for 500-PE systems with high loading and low discharge limits. CAPEX accounts for an average of 48% of total LCCs. OPEX accounts for 37%, energy for 9% and parts for 6%. Figure 4.2, in contrast, considers the same treatment scenario, but for 2000-PE systems. The average CAPEX is reduced to 42% and OPEX to 36%; energy is increased to 18% and the parts cost is reduced to 5%.

The increase in the energy percentage occurs because the scale economy rates are higher for the other LCC elements, i.e. the other operational cost elements show a greater reduction per capita with increasing scale. Most systems show some level of reduction in specific per capita energy use with an

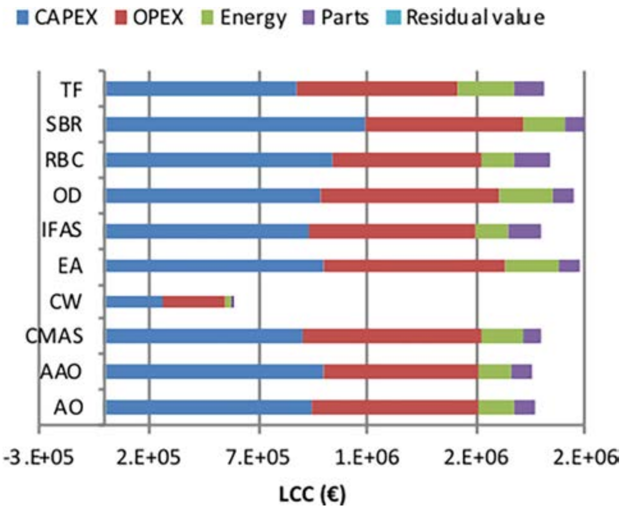


Figure 4.1. Life cycle cost distribution, 500 PE.

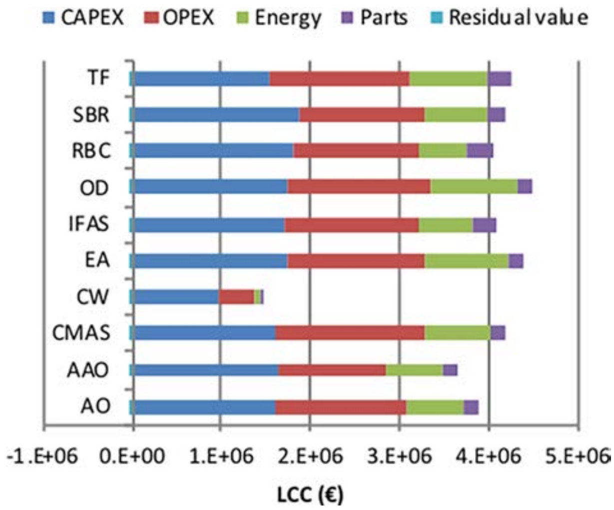


Figure 4.2. Life cycle cost distribution, 2000 PE.

increase in scale, e.g. the CMAS system's specific energy is reduced from 0.75 kWh/m^3 to 0.72 kWh/m^3 for an increase from 500 PE to 2000 PE. Because the discount rate used to calculate the energy UPV (5%) is greater than the OPEX UPV (3.5%), differences in the rate of change with respect to scale are increased. It is evident that, as WWTS scale becomes very small, CAPEX becomes the dominant LCC component for some systems. This trade-off between cost components puts the economic and environmental costs in direct conflict with each other because, if CAPEX becomes the basis for system comparison and selection, OPEX is a secondary consideration and it is the operational phase of a system's lifetime that contributes most to environmental impact.

4.1.2 Variation in organic load

A change in the variation in loading from low to high results in a collective average increase of all systems in OPEX (29–36%) and in energy (14–18%). CAPEX is a function of scale and, therefore, while the estimated CAPEX does not change with respect to changes in organic loading, the CAPEX percentage of the LCC decreases from 51% to 42%. The LCC distribution presented below (Figures 4.3 and 4.4) is based on sludge option 1, in which the cost of sludge disposal is minimal. In sludge option 2, where sludge disposal costs are at a maximum, OPEX is increased from 52% to 60% and energy from 9% to 11% and CAPEX is reduced from 35% to 25% of the total LCC.

4.1.3 Variation in discharge limits

Variation in discharge limits from most to least stringent, sludge option 1, results in a reduction in LCC of 20%, on average. Operational costs are reduced from 36% to 33% and energy from 18% to 13%, while CAPEX increases from 42% to 48%. Analysis of sludge disposal variation showed that drying beds had the lowest LCCs of the three options evaluated in all scenarios. The variation in LCC between options 1 and 3 ranged from 4% to 15%. The smallest difference in the values between options 1 and 3 – in which the LCC with the drying bed option is at its highest – occurs at small scales when organic loading is low, which reduces surface area requirement because drying bed surface area is a function of organic loading. Land is assumed not to lose its value and, therefore, systems with large surface areas have a greater residual value at the end of their lifetimes. The percentage difference in the systems' LCCs between options 1 and 2 ranged from 1% to 49%. Option 1 always yielded a lower LCC than option 2. The largest difference in values occurred at large scales, high loading and high limits, when solids retention times (SRTs) were at their lowest and sludge production at its highest.

4.1.4 Sludge treatment and disposal

The attached growth systems performed better in sludge option 2 than option 1 (Figures 4.5 and 4.6). The primary cause relates to the sludge dry solids concentration (DSC). Attached growth systems

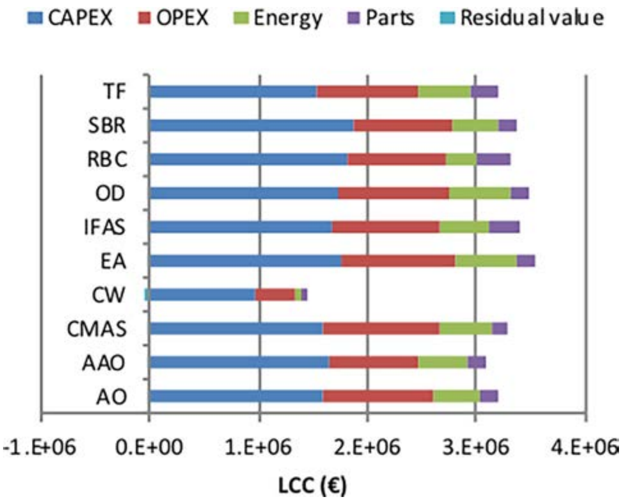


Figure 4.3. Life cycle cost distribution – low loading.

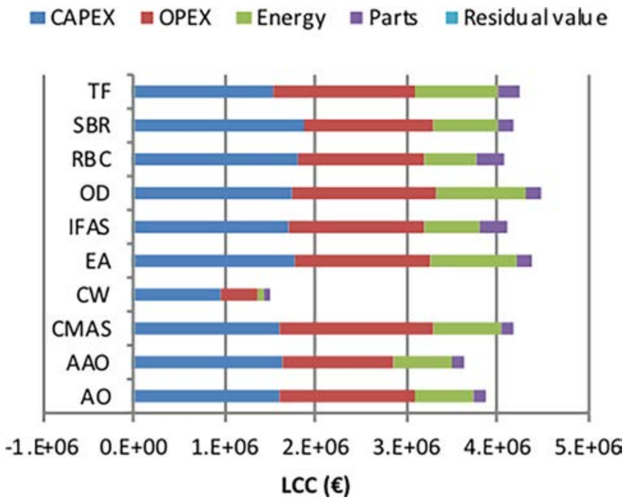


Figure 4.4. Life cycle cost distribution – high loading.

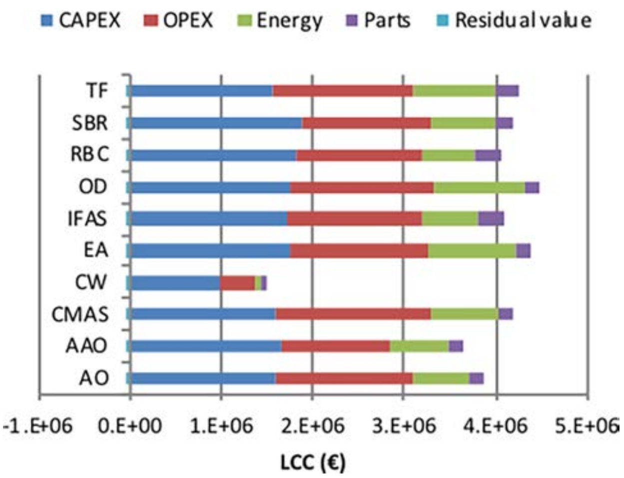


Figure 4.5. Life cycle cost distribution – sludge option 1.

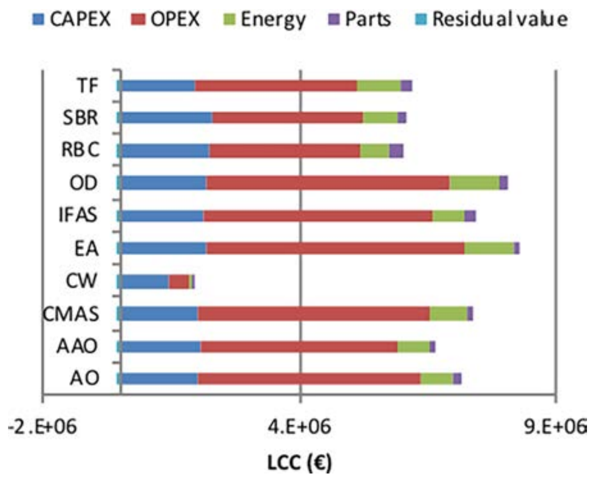


Figure 4.6. Life cycle cost distribution – sludge option 2.

generally produce higher-density sludge. TF sludge or humus DSC is reported to range from 1% to 4% (Turovskiy and Mathai, 2006). The average TF and RBC DSC value adopted for this study is 2.3%, whereas the value adopted for waste activated sludge (WAS) is 1.3%. Although the difference is small, the effect on sludge volume is significant. For a 2000-PE plant with high organic loading and phosphorus removal, the sludge mass was estimated to be 200 kg DS/day. Without any treatment, the TF system sludge volume is 5.06 m^3/day, and the AAO system sludge volume is 6.95 m^3/day. The difference of 1.89 m^3/day equates to an additional removal cost of €141.75/day, or €51,738/year, for disposal by an external contractor. Similarly, despite the higher CAPEX associated with the SBR system, the 4.3% sludge DSC value adopted (Janczukowicz *et al.*, 2001) resulted in the system outperforming other activated sludge-based systems in this sludge disposal category. Because the cost of disposal is dependent on volume (for any given destination), it could have been assumed that the option to dewater and land spread would result in much lower LCCs. Moreover, the specific cost of removal of sludge from the site for land spreading was 20% lower than using the external contractor at €60 and €75/m^3, respectively. However, for small scale systems with low loading and ammonia reduction requirements, the external contractor option becomes more economical when the specific cost of disposal falls below €65/m^3.

4.1.5 Life cycle impact assessment

The life cycle impact assessment methodology results presented below (Figures 4.7 to 4.12) represent scenarios with high loading and low discharge limits for 500–2000 PE in sludge option 1. These scenarios were chosen to provide a general overview because they include all systems considered. The outputs of each category have been normalised with Western Europe normalisation factors (2001–2013) for the purpose of comparison (Jolliet *et al.*, 2003). The CW systems had the lowest environmental impact in all impact categories and scenarios. Direct aerial emissions of CO_2 were, on average, 25% lower than for the electro-mechanical systems, but mitigation of a large proportion of the downstream CO_2 emissions from energy and chemical production increases this value to almost 50%. Other impact categories, such as AP and ADPf, show a much greater difference in output between CW and electro-mechanical systems, while categories such as EP and human toxicity potential (HTP) show similar ranges of output because the magnitude of impact in these categories is influenced more by direct emissions than by those generated by upstream processes.

The general trend observed in the environmental profiles of the electro-mechanical systems was that, as discharge limits became more stringent, the systems' EP levels decreased; however, in most of the other environmental impact categories, levels increased as the demand for energy and chemicals grew. This inverse relationship illustrates a trade-off between impact categories and leads to a wider debate on regional versus local impacts, which is warranted but beyond the scope of the current study.

The electro-mechanical systems' GWP and AP profiles varied with energy and chemicals use. The impact from energy use was found to be heavily influenced by the Irish electrical grid mix, of which 93% is made up of fossil-based fuels. Natural gas accounts for 43%, hard coal for 27.1%, heavy fuel oil for 14.8% and peat for 8.3% (Thinkstep GaBi, 2012). This produced an aggregated equivalent CO_2 potential of 0.58 kg CO_2/kWh produced. Comparing this with the 0.059 kg CO_2/kWh produced in Sweden, whose grid consists of 46% hydro and 46% nuclear (Thinkstep GaBi, 2012), it is evident that a system's GWP is subject to the quantity of energy used and the mode of national energy generation. AP is dominated by the impact from chemical production and, therefore, systems with the ability to mitigate a percentage of chemical use, such as the AAO system, have a more favourable AP profile (Figure 4.8). A simplification was used to estimate EP; as the discharge limits are considered to represent the final effluent concentrations of a given pollutant, most systems have the same EP output, with slight variations where there is EP from other sources. The main contributors to ADPf are energy and chemical production, which account for 22–60% and 40–78% of the total impact, respectively. As with other impact categories to which energy and chemicals are the main contributors, the AAO system has the lowest potential (Figure 4.10). Over 90% of HTP is attributed to the heavy metal content of sludge. The remaining 10% varies between energy and chemical production depending on variations in other parameters. The concentration of heavy metals was estimated as a percentage of the sludge DSC and, therefore, the HTP was largely consistent between systems (Figure 4.11). The most significant variation occurs between sludge disposal options. According to Stefanakis and Tsihrintzis (2012), on average, metal concentrations in the residual sludge in drying reed beds are about 30% of the original values. Most of the metals accumulate in the gravel layer (49%), while plant uptake accounts for 3% and 16% of heavy metals remain in the drained water. Therefore, sludge drying beds can provide a good option for reducing toxicity potential in categories in which sludge is the dominant contributor. Note that, from an LCA perspective, the heavy metal content in sludge is the major contributor to the LCA toxicity impact factor. It is also worth noting however that, according to the National Wastewater Sludge Management Plan,[9] the concentration of metals in Irish sludge is consistently low.

4.2 Performance Assessment Methodology

The results from piloting the performance assessment methodology, KPICalc, have been reported in an earlier EPA report (Fitzsimons *et al.*, 2016) and therefore will not be repeated in this report. It should be noted, however, that, in the case of small-scale WWTPs, additional care and attention are required

9 https://www.water.ie/projects-plans/our-plans/wastewater-sludge-management/Final-NWSMP.pdf

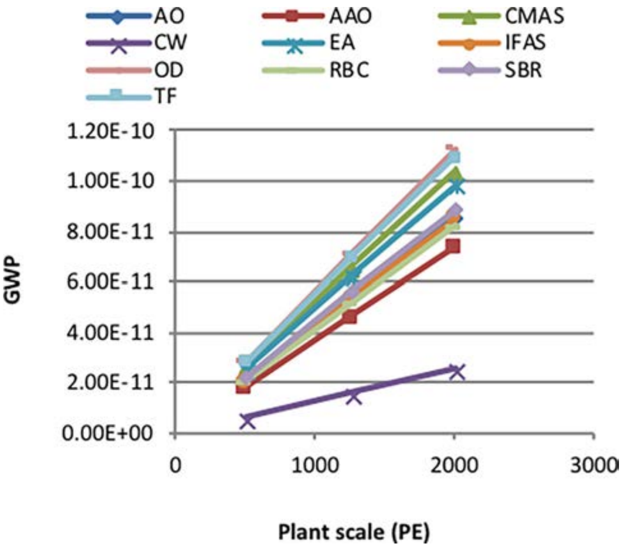


Figure 4.7. Global warming potential.

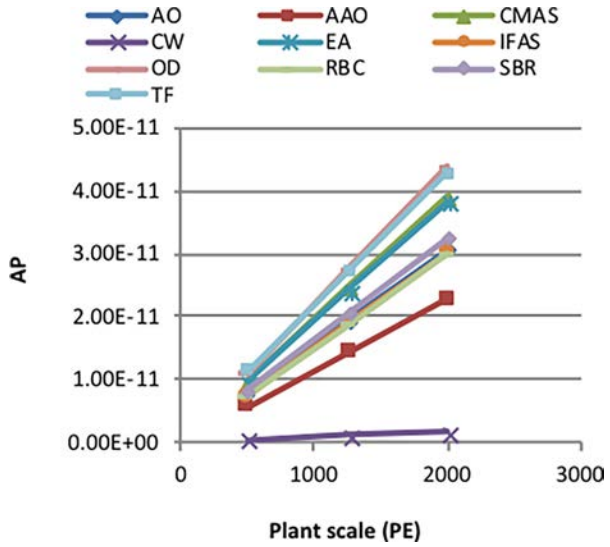


Figure 4.8. Acidification potential.

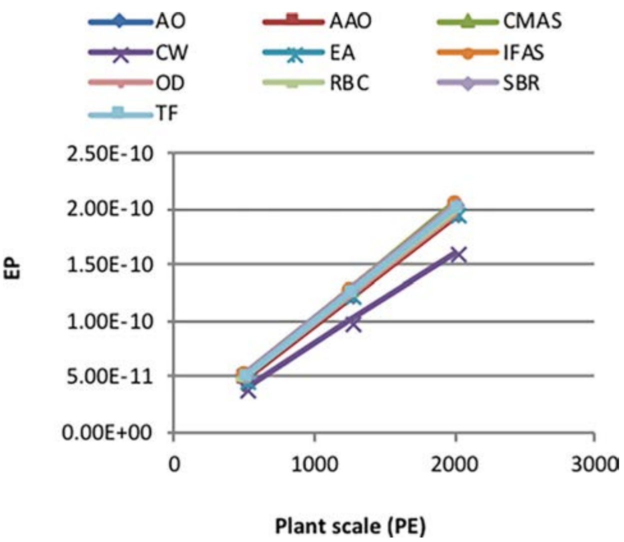


Figure 4.9. Eutrophication potential.

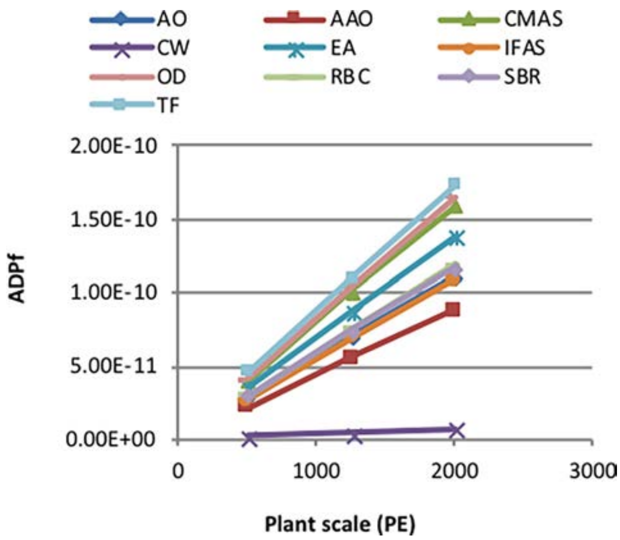


Figure 4.10. Abiotic resource depletion potential (fossil).

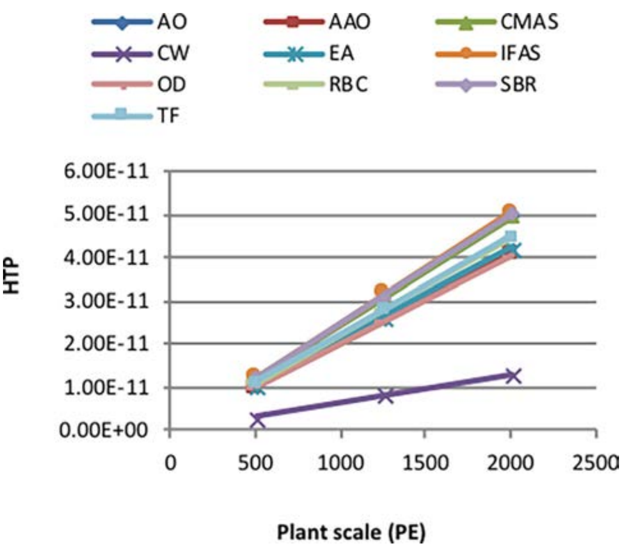


Figure 4.11. Human toxicity potential, sludge 1.

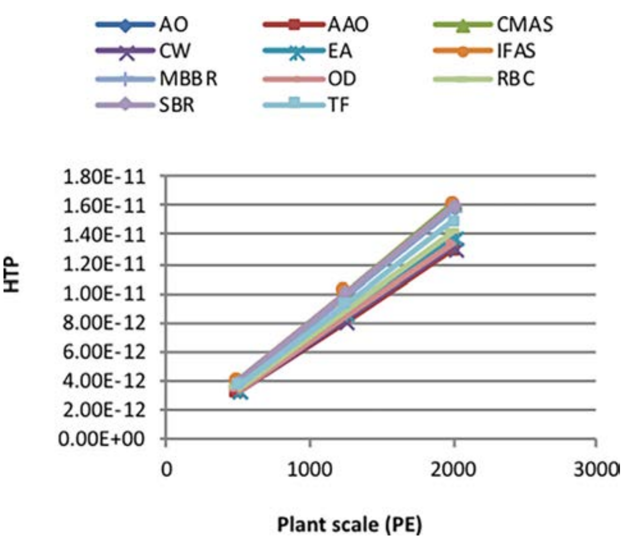


Figure 4.12. Human toxicity potential, sludge 3.

when undertaking performance assessment. Implementing KPICalc is achievable even if the available data are limited; however, it is recommended that caution is exercised if benchmarking such facilities against other WWTPs or drawing significant conclusions from the exercise. This reflects challenges previously identified regarding data management practices in Irish WWTPs.

It is recommended that managers of small-scale WWTPs who want to benchmark their WWTP against its peers undertake a period of intensive wastewater testing and data collection for performance assessment purposes. This ensures that sufficient results are provided, enabling adequate conclusions to be drawn from benchmarking.

4.3 Comparable Wastewater Treatment Plant Identification Tool

The comparable WWTP identifier tool was developed and piloted using the preloaded Irish WWTP data, which are supplied by each WWTP to the EPA for regulation purposes. The recommended identifiers (design capacity, cBOD, ammonium and orthophosphate), along with the recommended subdivisions, were selected for piloting.

4.3.1 Assessment of the spread of discharge limit concentrations

The combinations considered when developing this tool (which can be changed by the user) are summarised in Table 4.3. For example, there were 66 WWTPs with a PE of 10,000 or more and 152 WWTPs with an ammonium discharge limit of between 1 mg/l and 5 mg/l.

Table 4.4 condenses the results obtained from the comparable WWTP identifier tool when using the above identifiers and subdivisions. Several well-populated combinations of discharge limit identifiers and subdivisions can be identified in Table 4.4. These combinations are spread across both stringent (e.g. cBOD 5.01–10 mg/l, ammonium 0–1 mg N/l and orthophosphate 0–1 mg P/l) and moderately stringent discharge limit concentrations (e.g. cBOD 10.01–25 mg/l, ammonium 1.01–5 mg N/l and orthophosphate 1.01 mg/l or more), such that there are WWTPs of each size category that have the above combinations of discharge limits.

With more WWTPs becoming subject to increasingly stringent discharge limits, greater focus must be given to improving WWTP performance in a sustainable manner, as stringent discharge limit concentrations typically require increased WWTP contaminant removal performance, resulting in additional resource consumption. Benchmarking of WWTPs can be more effective where there are a number of WWTPs within a given category (Table 4.4); although some WWTPs share similar subdivisions, many are unique in terms of their discharge limit concentrations. Therefore, minor changes to an discharge limit could move a WWTP into a category where it can be compared with others.

4.3.2 Comparing wastewater treatment plants as part of KPICalc result analysis

Using the WWTP identifier tool, users can select comparable WWTPs for benchmarking using KPICalc results. If required, users can then assess the percentage of design capacity utilised by using KPICalc results to ensure that the WWTPs are using similar levels of their design capacity; WWTPs may differ substantially in terms of design capacity

Table 4.3. Distribution of preloaded WWTP data across identifiers (PE, cBOD, etc.) and subdivisions (groupings considered: < 500 PE, etc.)

Design Capacity (PE)	Number of WWTPs	cBOD (mg/l)	Number of WWTPs	Ammonium (mg N/l)	Number of WWTPs	Orthophosphate (mg/l)	Number of WWTPs
< 500	36	≤ 5	15	≤ 1	72	≤ 1	147
500–1000	94	> 5–10	69	> 1–5	152	> 1	80
1001–2000	72	> 10–25	235	> 5	34		
2001–10,000	87	> 25	2				
> 10,000	66						

Table 4.4. Condensed results from the application of the comparable WWTP identifier tool

cBOD (mg/l)	Ammonium (mg/l)	Orthophosphate (mg/l)	Total numbers of WWTPs by design capacity				
			< 500 PE	500–1000 PE	1001–2000 PE	2001–10,000 PE	> 10,000 PE
–	–	–	10	12	3	5	4
0–5	–	–	0	1	0	0	0
0–5	0–1	–	0	0	0	1	0
0–5	–	0–1	0	0	0	0	1
0–5	0–1	0–1	0	1	3	3	1
0–5	0–1	≥1.01	1	0	0	0	0
0–5	1.01–5	0–1	0	1	1	1	0
5.01–10	–	–	0	5	2	1	0
5.01–10	0–1	–	0	3	0	1	1
5.01–10	1.01–5	–	0	0	1	0	0
5.01–10	–	0–1	0	0	1	1	0
5.01–10	0–1	0–1	4	6	9	11	6
5.01–10	1.01–5	0–1	1	3	5	4	0
5.01–10	1.01–5	≥1.01	0	1	1	0	1
5.01–10	5.01+	≥1.01	0	1	0	0	0
10.01–25	–	–	6	8	8	4	12
10.01–25	0–1	–	0	0	1	1	1
10.01–25	1.01–5	–	1	0	1	7	9
10.01–25	5.01+	–	1	1	2	7	8
10.01–25	–	0–1	0	0	1	1	1
10.01–25	–	≥1.01	0	5	0	1	2
10.01–25	0–1	0–1	0	4	4	7	1
10.01–25	0–1	≥1.01	0	0	1	1	0
10.01–25	1.01–5	0–1	6	12	12	19	10
10.01–25	1.01–5	≥1.01	5	23	11	10	6
10.01–25	5.01+	0–1	0	3	2	0	1
10.01–25	5.01+	≥1.01	1	4	1	1	1
25.01+	1.01–5	≥1.01	0	0	2	0	0

utilisation, which can affect WWTP comparability. Similarly, discharge limit compliance results for each WWTP should be assessed to ensure that WWTPs are meeting these regulatory requirements (this is an initial output of KPICalc).

To further check WWTP comparability, users may wish to assess additional WWTP characteristics to differentiate between WWTPs within a comparable group. This may be achieved by analysis of the KPICalc results as part of an optional secondary comparable WWTP identification step. Secondary identification of comparable WWTPs involves taking the WWTPs of interest identified during primary identification and comparing contextual information and KPI results using KPICalc outputs. If KPICalc results are being benchmarked across two comparable groups (e.g. in the case of a group containing only one WWTP), users must benchmark KPI results with caution and assess the contextual information provided in the survey module of KPICalc to aid in the identification of WWTP characteristics that may affect comparability.

It should also be noted that the cost of achieving compliance has not yet been widely analysed, and this should be considered. Such analyses should include the additional energy costs and carbon emissions associated with achieving particularly stringent discharges and compared with the potential benefits of these limits. These issues are central to the LCCA and LCA components of this research.

4.3.3 Statistical agreement of wastewater quality sampling methods

The numerical results obtained from Lin's CCC provide a more reliable method for assessing method agreement by analysing each pair of reported results and calculating an overall indicator of agreement. Lin's CCC outputs a numerical value between –1 and +1 (similar to correlation values), with a CCC of 1 indicating perfect agreement and a CCC of –1 indicating perfect disagreement. In practice, a degree of error is often present and uncontrollable; therefore, it is rare to find perfect agreement (CCC = 1) between two methods. Table 4.5 presents the results obtained from statistical agreement analysis using Lin's CCC. The precision and bias correction factors are also presented.

From analysis of the CCC results, there was insufficient agreement between any of the selected grab sampling methods and the flow-paced method in calculating daily mean concentrations in both WWTP A and WWTP B (Table 4.5). Poor agreement between the M6G method (the mean of six grab samples collected over 24 hours) and FP method (a 24-hour flow-paced composite sample) can be seen in Table 4.5, with the highest CCC result obtained by reporting the daily mean influent ammonium concentration using the M6G and FP methods in WWTP B (CCC = 0.6705).

In the M3WDG (mean of three grab samples collected during working day) and M3RG (mean of three randomly selected grab samples) methods, three grab samples were used to calculate daily mean concentrations to remove the effect of the number of

Table 4.5. Agreement between flow-paced sampling and various grab sampling frequencies

Parameter	Sampling methods	Sample size (days)	CCC	95% confidence interval	Pearson ρ (precision)	Bias correction factor C_b (accuracy)
WWTP A						
COD	M3WDG and FP	28	0.2520	0.006437 to 0.4689	0.3912	0.6442
	M3RG and FP	32	0.0465	–0.09497 to 0.1861	0.1178	0.3946
	M6G and FP	26	0.1743	–0.03651 to 0.3702	0.3268	0.5332
TSS	M3WDG and FP	28	0.4749	0.1826 to 0.6901	0.548	0.8666
	M3RG and FP	32	0.4823	0.2524 to 0.6606	0.6329	0.762
	M6G and FP	26	0.5800	0.2667 to 0.7824	0.5909	0.9815
Ammonium	M3WDG and FP	28	0.2662	–0.004365 to 0.5004	0.3746	0.7105
	M3RG and FP	32	0.2534	0.04620 to 0.4397	0.4389	0.5774
	M6G and FP	26	0.4851	0.2378 to 0.6733	0.6815	0.7117
Orthophosphate	M3WDG and FP	26	0.3076	0.05360 to 0.5242	0.4756	0.6467
	M3RG and FP	31	0.1031	–0.06378 to 0.2643	0.2276	0.4529
	M6G and FP	24	0.2607	0.06078 to 0.4405	0.5488	0.475
WWTP B						
COD	M3WDG and FP	30	0.3473	0.08229 to 0.5665	0.4497	0.7723
	M3RG and FP	30	0.5336	0.2903 to 0.7120	0.6308	0.8459
	M6G and FP	29	0.4155	0.1922 to 0.5979	0.5907	0.7035
TSS	M3WDG and FP	30	0.3639	0.01744 to 0.6324	0.3696	0.9848
	M3RG and FP	30	0.3852	0.03578 to 0.6507	0.3852	1.0000
	M6G and FP	29	0.4475	0.1257 to 0.6840	0.4699	0.9523
Ammonium	M3WDG and FP	30	0.6485	0.3954 to 0.8101	0.6681	0.9706
	M3RG and FP	30	0.6358	0.3689 to 0.8058	0.6489	0.9799
	M6G and FP	29	0.6705	0.4362 to 0.8196	0.7069	0.9485
Orthophosphate	M3WDG and FP	30	0.3892	0.06330 to 0.6401	0.4108	0.9475
	M3RG and FP	30	0.2976	–0.05637 to 0.5851	0.3036	0.9801
	M6G and FP	28	0.5101	0.2257 to 0.7143	0.5575	0.9149

M3WDG, mean of three grab samples collected during working day (8 a.m., 12 p.m. and 4 p.m.); M3RG, mean of three randomly selected grab samples; M6G, mean of six grab samples collected over 24 hours; FP, a 24-hour flow-paced composite sample, TSS, total suspended solids.

samples used in each method. From analysis of the CCC results, although identical numbers of samples are used in each method, the timing of these samples influenced each method's agreement with flow-paced sampling, which is connected to daily influent variability. CCC results for the M3WDG and M3RG methods are seen to vary across the four pollutants of interest and in both WWTPs; however, the strength of agreement of these methods remain poor.

Overall, it is clear that, in the WWTPs analysed, flow-paced sampling would be the most representative method of sampling wastewater of all the methods presented in this study. This is to be expected in the case of WWTPs dealing with large daily variations in influent characteristics and influent pollutant concentrations. However, it should be noted that some WWTPs may show low influent variability; in this case, a small number of grab samples may be as representative of the daily mean as flow-paced samples.

4.3.4 Application of the statistical agreement analysis methods in Irish WWTPs

Irish WWTPs with a design capacity greater than 2000 PE are required by regulation to provide flow-paced composite sampling facilities; however, WWTPs can use either time-paced or flow-paced composite sampling for regulatory monitoring (which typically takes place on a monthly basis). Given the cost associated with maintaining flow-paced sampling facilities, it may be unfeasible to provide this facility on the influent and effluent stream in WWTPs that have a design capacity of less than 2000 PE and at the frequency required for performance assessment. Therefore, it may be necessary to (1) identify alternative representative sampling methods for influent and effluent sampling or (2) prioritise the provision of flow-paced sampling on either the influent or the effluent stream (whichever presents the largest variation in pollutant concentrations and flow rates) and opt for a lower cost but representative method of sampling on the remaining stream.

The processes used in the WWTPs can also present a buffering capacity, which (1) facilitates influent wastewater mixing (normally via a balance tank at or close to the influent works) and (2) potentially regulates the flow rate at which the effluent is discharged. Therefore, accounting for flow variation may not be as critical in effluent sampling as in influent sampling. Therefore, for WWTP operations purposes, where it is necessary to prioritise flow-paced provision on one wastewater stream, it may be more economically feasible to utilise flow-paced sampling on the influent stream. Subsequently, using the methods presented in this chapter, it is crucial that a grab sampling or time-paced sampling methodology with a high statistical strength of agreement with flow-paced methods is identified for the effluent stream.

5 Conclusions

Two software tools were developed during this and earlier affiliated research (Fitzsimons *et al.*, 2016): DST and KPICalc.

There is no "one size fits all" wastewater treatment design solution for small systems. Variations in plant scale, organic and hydraulic loading, discharge limits and land availability can influence the economic and environmental performance of a treatment system to the extent that its suitability for a given location may be less than that of competing systems; therefore, system applicability should be assessed on a case-by-case basis.

Using the DST tool developed as part of this research and the previously discussed user inputs, CWs were found to be a viable alternative to conventional electro-mechanical systems in locations where land availability at a reasonable cost is not an issue. In addition, CWs were found to have a more favourable environmental profile as a result of reduced energy and chemical requirements.

Energy use contributes significantly to both the operational cost and the environmental impact of small electro-mechanical WWTSs. Small agglomerations do not benefit from energy scale economies and incur a higher per capita energy cost. This places even greater importance on understanding the specific energy costs associated with a given system in a given location.

The environmental impact resulting from energy consumption is influenced by both the quantity of energy used and the country's electrical grid mix. Improvements in WWTS efficiencies and operation, coupled with the integration of more renewable sources of energy into the national electrical grid mix, will improve a system's environmental profile.

There are significant trade-offs between regional and global environmental impact categories. This is largely dictated by a plant's discharge limits, i.e. more stringent discharge limits will reduce regional impact (eutrophication, toxicity) while increasing global impact (global warming, acidification).

Of the on-site sludge treatment options considered in this study, drying beds have the greatest potential to reduce terrestrial toxicity because of their capacity to reduce heavy metal concentrations. However, as with CWs, the large surface areas required for drying beds may restrict their selection feasibility. Furthermore, in more suburban locations, there may be a social aspect to consider because of odour-related issues.

Great care should be taken when selecting WWTPs against which to benchmark. WWTP size and regulatory requirements can affect resource consumption and subsequent WWTP performance benchmarking. In addition, the complex relationship that is often seen between these WWTP characteristics results in the need for various characteristics to be assessed simultaneously.

The WWTP grouping methodology (comprising a DST to facilitate rapid identification of comparable WWTPs) has shown great potential in the identification of WWTPs that can be compared and benchmarked using KPICalc results. In addition, the comparable WWTP identification tool offers the advantage of rapid identification of WWTPs and the ability to repeat the analysis with ease, when compared with any manual method of identification.

By assessing the statistical agreement between the various sampling methodologies in a WWTP, it may be possible to identify if a representative method of wastewater sampling for benchmarking processes that is low cost and low maintenance (and therefore more suited to the long-term sampling associated with daily benchmarking) can replace costlier methods, such as flow-paced composite sampling. Selecting a reliable, yet low-maintenance, method may potentially mitigate current data availability and accuracy issues.

6 Recommendations

6.1 Assess Performance Using Multiple Criteria and KPIs over the System Life Cycle

The economic and environmental costs of WWTPs are a function of many variables, including some, such as influent concentrations and discharge requirements, over which a plant manager has little control. WWTP performance should be assessed holistically using multiple criteria over the entire system life cycle. Compliance with discharge limits can increase energy and chemical demands, which can produce unintended consequences, such as an increase in global warming potential. A more in-depth analysis will be required to assess future trade-offs between environmental compartments as discharge limits become more strict and new limits are introduced to deal with emerging pollutants. Benchmarking of WWTPs should be further developed as a performance optimisation and management tool. This will require, in particular, the use of flow-paced sampling at WWTPs. Methodologies to determine minimal data requirements for effectively implementing performance management, life cycle assessment and benchmarking tools should be considered.

6.2 Performance Assessment Methodology Application

The performance assessment and comparable WWTP identification methodologies developed in this study can be applied to standalone software applications and existing technologies such as building information modelling, digital asset management and supervisory control and data acquisition. By applying these methodologies to existing technologies, the need for additional training on new software applications is minimised and an opportunity for greater collaboration between personnel in the management and optimisation of WWTPs is presented.

6.3 Sampling Methods at WWTPs

The use of flow-paced sampling is likely to result in the most accurate representation of influent and effluent contaminant loads. However, this study presents a methodology whereby the use of carefully timed grab sampling can be analysed and may result in sufficiently accurate sampling, particularly of effluent streams.

Lin’s CCC and other statistical agreement assessment methods have been used in many epidemiological studies to assess agreement between medical methods. However, such statistical analysis methods have not been widely applied throughout the water and wastewater sector. This report found that Lin’s CCC can be readily and practically applied for the assessment of wastewater quality sampling methods; it is recommended that the strength-of-agreement values proposed by McBride (2007) be adapted to suit the objectives of the study.

6.4 System Selection

Ideally, all WWTS selection would be based on sustainable, socially acceptable designs with low environmental impact and economic cost. Historically, CAPEX has been the dominant consideration and other criteria may have been overlooked. It is therefore recommended that economic cost evaluations and comparisons are carried out over the lifetime of the systems using appropriate cost models, such as those presented in this report. A separate PV estimate should be used for energy to assess the sensitivity of LCC to variations in energy discount rates, and this should be re-evaluated regularly as energy prices fluctuate.

Environmental impact should be another important design criterion, particularly in situations where the LCCs of competing systems are within a margin of uncertainty. Based on the DST model developed during this research and the input data used:

- DSTs, such as that presented in this report, can (or should) be used to compare the relative merits of both natural and electro-mechanical WWTSs on a case-by-case basis.
- Sludge disposal costs are location-specific and alternative disposal options should be assessed during LCCA for competing systems.

7 Publications Arising from This Research

7.1 Articles

Doherty, E., Fitzsimons, L., Corcoran, B., Delaure, Y. and Clifford, E., 2017. A resource consumption benchmarking system for WWTPs. *Asian Water Magazine* Issue 12: 13–16.

McNamara, G., Fitzsimons, L., Doherty, E. and Clifford, E., 2017. The evaluation of technologies for small, new design, wastewater treatment systems. *Journal of Desalination and Water* 91: 12–22.

McNamara, G., Horrigan, M., Phelan, T., Fitzsimons, L., Delaure, Y., Corcoran, B., Doherty, E. and Clifford, E., 2016. Life cycle assessment of waste water treatment plants in Ireland. *Journal of Sustainable Development of Energy, Water and Environment Systems* 4(3): 216–233.

Fitzsimons, L., Horrigan, M., McNamara, G., Doherty, E., Phelan, T., Corcoran, B., Delauré, Y. and Clifford, E., 2016. Assessing the thermodynamic performance of Irish municipal wastewater treatment plants using exergy analysis: a potential benchmarking approach. *Journal of Cleaner Production* 131: 387–398.

Doherty, E., McNamara, G., Fitzsimons, L. and Clifford, E., 2014. Design and implementation of a resource consumption benchmarking methodology cognisant of data accuracy for Irish wastewater treatment plants. *Journal of Cleaner Production* 165: 1529–1541.

7.2 Peer-reviewed Conference Papers

Doherty, E., Fitzsimons, L., McNamara, G. and Clifford, E., 2017. Resource benchmarking: grouping Irish wastewater treatment plants using discharge licence data. Paper presented at Pi 2017 – Specialist Conference on Benchmarking and Performance Assessment, 15–17 May 2017, Vienna, Austria, pp. 15–17.

Doherty, E., Fitzsimons, L., McNamara, G. and Clifford, E., 2017. Assessment of the statistical agreement between wastewater sampling methods for daily wastewater treatment plant performance benchmarking purposes. Paper presented at Pi 2017 – Specialist Conference on Benchmarking and Performance Assessment, 15–17 May 2017, Vienna, Austria.

McNamara, G., Fitzsimons, L., Doherty, E. and Clifford, E., (2016) The evaluation of technologies for small, new design, wastewater treatment systems. Paper presented at the 13th IWA Specialised Conference on Small Water and Wastewater Systems and 5th IWA Specialised Conference on Resource-orientated Sanitation, 14–16 October 2016, Athens, Greece.

Doherty, E., McNamara, G., Phelan, T., Horrigan, M., Fitzsimons, L., Corcoran, B., Delaure, Y. and Clifford, E., 2015. Benchmarking resource efficiency in wastewater treatment plants: developing best practices. Paper presented at the IWA International Conference on Water Efficiency and Performance Assessment of Water Services, 20–24 April 2015, Cincinnati, OH, USA.

McNamara, G., Fitzsimons, L., Doherty, E., Clifford, E. and Delaure, Y., 2015. Performance metrics in life cycle assessments of wastewater treatment plants. Paper presented at the 10th Conference on Sustainable Development of Energy, Water and Environment Systems, 27 September to 2 October 2015, Dubrovnik, Croatia.

Doherty, E., Fitzsimons, L., Corcoran, B., Delaure, Y. and Clifford, E., 2014. Design and implementation of a resource consumption benchmarking system for wastewater treatment plants. Paper presented at the IWA Water, Energy and Climate Conference 2014, 21–23 May 2014, Mexico City, Mexico.

McNamara, G. Horrigan, M., Phelan, T., Fitzsimons, L., Delaure, Y., Doherty, E. and Clifford, E., 2014. Life cycle assessment of waste water treatment plants in Ireland. Paper presented at the 1st South East European Conference on Sustainable Development of Energy, Water and Environment Systems (SEE SDEWES Ohrid 2014), 29 June to 3 July 2014, Ohrid, Republic of Macedonia.

7.3 Invited Talks

Fitzsimons, L., McNamara, G., Phelan, T., Horrigan, M., Doherty, E., Clifford, E., Delaure, Y. and Corcoran, B. 2014. Energy and water: wastewater treatment. Water: the greatest global challenge. 2nd Annual Conference, 27–28 November 2014, DCU Water Institute, Dublin.

7.4 Theses

McNamara, G., 2017, Economic and environmental cost assessment of wastewater treatment systems: a life cycle perspective. PhD thesis. Dublin City University.

Doherty, E., 2017, Development of new benchmarking systems for wastewater treatment facilities. PhD thesis. National University of Ireland, Galway. Available online: https://aran.library.nuigalway.ie/handle/10379/6637

References

Arditi, D. and Messiha, H.M. 1999, Life cycle cost analysis (LCCA) in municipal organizations. *Journal of Infrastructure Systems* 5(1): 1–10.

Babatunde, A., Zhao, Y., O'Neill, M. and O'Sullivan, B., 2008. Constructed wetlands for environmental pollution control: a review of developments, research and practice in Ireland. *Environment International* 34(1): 116–126.

Belmont, M.A., Cantellano, E., Thompson, S., Williamson, M., Sánchez, A. and Metcalfe, C.D., 2004. Treatment of domestic wastewater in a pilot-scale natural treatment system in central Mexico. *Ecological Engineering* 23(4): 299–311.

Bolles, S., 2006. Modeling wastewater aeration systems to discover energy savings opportunities. Process Energy Services, LLC, Londonderry, NH.

Cabrera, E., Dane, P., Haskins, S. and Theuretzbacher-Fritz, H., 2009. Benchmarking revisited: the global search for performance improvement. *American Water Works Association Journal* 101(11): 22–26.

Carlson, S.W. and Walburger, A., 2007. *Energy Index Development for Benchmarking Water and Wastewater Utilities*. American Water Works Association, Denver, CO.

Corominas, L., Foley, J., Guest, J.S., Hospido, A., Larsen, H.F., Morera, S. and Shaw, A., 2013. Life cycle assessment applied to wastewater treatment: state of the art. *Water Research* 47(15): 5480–5492.

Davis, M.L., 2010. *Water and Wastewater Engineering.* McGraw-Hill, New York, NY, USA.

Dhillon, B.S., 2009. *Life Cycle Costing for Engineers.* CRC Press, Boca Raton, FL, USA.

Eisenberger, I. and Lorden, G., 1977. Life cycle costing: practical considerations. *DSN Progress Report* 42(40): 102–109.

EPA (Environmental Protection Agency), 2015. *Urban Waste Water Treatment in 2015*. EPA, Johnstown Castle, Ireland.

EPA (Environmental Protection Agency), 2016. *Ireland's Environment – An Assessment 2016, Environment and the Economy: Overview*. EPA, Johnstown Castle, Ireland.

Fitzsimons, L., Clifford, E., McNamara, G., Doherty, E., Phelan, T., Horrigan, M., Delauré, Y. and Corcoran, B., 2016. *Increasing Resource Efficiency in Wastewater Treatment Plants*. Environmental Protection Agency, Johnstown Castle, Ireland.

Foess, G.W., Steinbrecher, P., Williams, K. and Garrett, G., 1998. Cost and performance evaluation of BNR processes, *Florida Water Resources Journal* 11: 11–13.

Foladori, P., Vaccari, M. and Vitali, F. 2015. Energy audit in small wastewater treatment plants: methodology, energy consumption indicators, and lessons learned. *Water Science and Technology* 72(6): 1007–1015.

Gallego, A., Hospido, A., Moreira, M.T. and Feijoo, G., 2008. Environmental performance of wastewater treatment plants for small populations. *Resources, Conservation and Recycling* 52(6): 931–940.

Garrido-Baserba, M., Sobhani, R., Asvapathanagul, P., McCarthy, G.W., Olson, B.H., Odize, V., Al-Omari, A., Murthy, S., Nifong, A. and Godwin, J., 2016. Modelling the link amongst fine-pore diffuser fouling, oxygen transfer efficiency, and aeration energy intensity. *Water Research* 111: 127–139.

Gkika, D., Gikas, G.D. and Tsihrintzis, V.A., 2014. Construction and operation costs of constructed wetlands treating wastewater. *Water Science and Technology* 70(5): 803–810.

Gratziou, M.K, Tsalkatidou, M. and Kotsovinos, N.E., 2006. Economic evaluation of small capacity sewage processing units. *Global NEST Journal* 8(1): 52–60.

Guinée, J.B., Gorrée, M., Huppes, G., Klein, R., de Koning, A., van Oers, L., Sleeswijk, A.W., Suh, S., and Udo de Haes, H.A., 2001. *Handbook on Life Cycle Assessment – Operational Guide to the ISO standards.* Kluwer Academic Publishers, Dordrecht, the Netherlands.

Heino, O.A., Takala, A.J. and Katko, T.S., 2011. *Challenges to Finnish Water and Wastewater Services in the Next 20–30 Years, Tampere, Finland*.

Henze, M., 2008. *Biological Wastewater Treatment: Principles, Modelling and Design*. IWA Publishing, London.

Hospido, A., Moreira, M. and Feijoo, G., 2008. A comparison of municipal wastewater treatment plants for big centres of population in Galicia (Spain). *The International Journal of Life Cycle Assessment* 13(1): 57–64.

Irish Environment, 2016. *Life Cycle Assessment*. Available online: http://www.irishenvironment.com/iepedia/life-cycle-assessment/ (accessed 13 October 2016).

ISO (International Organization for Standardization), 1997. *ISO 14040: Environmental Management – Life Cycle Assessment – Principles and Framework*. ISO, Brussels.

ISO (International Organization for Standardization), 1998. *ISO 14041: Environmental Management – Life Cycle Assessment – Goal and Scope Definition and Inventory Analysis.* ISO, Brussels.

ISO (International Organization for Standardization), 2000. *ISO 14042: Environmental Management – Life Cycle Assessment – Life Cycle Impact Assessment*. ISO, Brussels.

Janczukowicz, W., Szewczyk, M., Krzemieniewski, M. and Pesta, J., 2001. Settling properties of activated sludge from a sequencing batch reactor (SBR). *Polish Journal of Environmental Studies* 10(1): 15–20.

Johnson, T., Steichen, M., Shaw, A., McQuarrie, J. and Hunter, G., 2006. When is IFAS the right choice? *Proceedings of the Water Environment Federation* 2006(13) 200–224.

Johnson, T.L., McQuarrie, J.P. and Shaw, A.R., 2004. Integrated fixed-film activated sludge (IFAS): the new choice for nitrogen removal upgrades in the United States. *Proceedings of the Water Environment Federation* 2004(16): 296–318.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G. and Rosenbaum, R., 2003. IMPACT 2002: a new life cycle impact assessment methodology. *The International Journal of Life Cycle Assessment* 8(6): 324.

Jones, G.M., Bosserman, B.E., Sanks, R.L. and Tchobanoglous, G. 1989. *Pumping Station Design*. Butterworth Publishers, Waltham, MA.

Joyce, M. and Carney, C., 2012. *Management Options for the Collection, Treatment and Disposal of Sludge Derived from Domestic Wastewater Treatment Systems.* Strive Report 2012-W-DS-9. Environmental Protection Agency, Johnstown Castle, Ireland.

Kalbar, P.P., Karmakar, S. and Asolekar, S.R., 2013. Assessment of wastewater treatment technologies: life cycle approach. *Water and Environment Journal* 27(2): 261–268.

Kayranli, B., Scholz, M., Mustafa, A., Hofmann, O. and Harrington, R., 2010. Performance evaluation of integrated constructed wetlands treating domestic wastewater. *Water, Air, & Soil Pollution* 210(1–4): 435–451.

Lecouls, H., 1999. ISO 14043: Environmental management – lifecycle assessment – life cycle interpretation. *International Journal of Life Cycle Assessment* 4: 254.

Lens, P., Zeeman, G. and Lettinga, G., 2001. *Decentralised Sanitation and Reuse,* IWA Publishing, New York, NY, USA.

Lindtner, S., Schaar, H. and Kroiss, H., 2008. Benchmarking of large municipal wastewater treatment plants treating over 100,000 PE in Austria. *Water Science and Technology* 57(10): 1487–1493. Available online: http://www.ncbi.nlm.nih.gov/pubmed/18520003 (accessed 27 November 2013).

LMI (Logistics Management Institute), 1965. *Life Cycle Costing in Equipment Procurement*. Washington, DC, USA.

Longo, S., d'Antoni, B.M., Bongards. M., Chaparro, A., Conrath, A., Fatone, F., Lema, J.M., Iglesias, M.M., Soares, A. and Hospido, A., 2016. Monitoring and diagnosis of energy consumption in wastewater treatment plants. A state of the art and proposals for improvement. *Applied Energy* 179: 1251–1268.

Lundin, M., Bengtsson, M. and Sverker, M., 2000. Life cycle assessment of wastewater systems: influence of system boundaries and scale on calculated environmental loads. *Environmental Science & Technology* 34(1): 180–186.

McBride, G.B., 2007. *Equivalence Measures for Comparing the Performance of Alternative Methods for the Analysis of Water Quality Variables*. National Institute of Water & Atmospheric Research Ltd, Auckland.

McEntee, D., 2006. *Constructed Wetlands: Opportunities for Local Authorities*. Dublin City Council, Dublin, Ireland.

Machado, A.P., Urbano, L., Brito, A., Janknecht, P., Salas, J. and Nogueira, R., 2007. Life cycle assessment of wastewater treatment options for small and decentralized communities. *Water Science and Technology* 56(3): 15–22.

McNamara, G., 2017. Economic and environmental cost assessment of wastewater treatment systems: a life cycle perspective. PhD thesis. Dublin City University, Dublin.

Martin, C. and Vanrolleghem, P.A., 2014. Analysing, completing, and generating influent data for WWTP modelling: A critical review. *Environmental Modelling and Software* 60: 188–201.

Matos, R., Cardoso, A., Ashley, R.M., Duarte, P., Molinari, A. and Schulz, A., 2003. *Performance Indicators for Wastewater Services*. IWA Publishing, London.

Metcalf & Eddy, 2014. *Wastewater Engineering: Treatment and Resource Recovery.* International Edition. McGraw-Hill, New York, NY, USA.

Mizuta, K. and Shimada, M., 2010. Benchmarking energy consumption in municipal wastewater treatment plants in Japan. *Water Science and Technology* 62(10): 2256–2262. Available online: http://www.ncbi.nlm.nih.gov/pubmed/21076210 (accessed 27 November 2013).

Molinos-Senante, M., Gómez, T., Caballero, R., Hernández-Sancho, F. and Sala-Garrido, R., 2015. Assessment of wastewater treatment alternatives for small communities: An analytic network process approach. *Science of The Total Environment* 532: 676–687.

NEIWPCC, 2008.*The Northeast Guide for Staffing at Publicly and Privately Owned Wastewater Treatment Plants*. NEIWPCC, Lowell, MA, USA.

Olsson, G., 2012. *Water and Energy: Threats and Opportunities.* IWA Publishing, New York, NY, USA.

O'Reilly, E., Clifford, E., Rodgers, M. and O'Donoghue, P., 2012. *Treatment and Monitoring of Nutrients, Odour and Sludge at a Small-Town Demonstration Wastewater Treatment System*. Strive Report 2006-ET-LS-12-M3. Environmental Protection Agency, Johnstown Castle, Ireland.

Parena, R. and Smeets, E., 2001. Benchmarking initiatives in the water industry. *Water Science and Technology* 44(2–3): 103–110.

Poloski, A.P., Bonebrake, M.L., Casella, A.M., Johnson, M.D., Toth, J.J., Adkins, H.E., Chun, J., Denslow, K.M., Luna, M. and Tingey, J.M. 2009. *Deposition Velocities of Non-Newtonian Slurries in Pipelines: Complex Simulant Testing*. Pacific Northwest National Laboratory, Richland, WA.

Pretel, R., Robles, A., Ruano, M., Seco, A. and Ferrer, J., 2016. Economic and environmental sustainability of submerged anaerobic MBR-based (AnMBR-based) technology as compared to aerobic-based technologies for moderate-/high-loaded urban wastewater treatment. *Journal of Environmental Management* 166: 45–54.

Rawal, N. and Duggal, S., 2016. Life cycle costing assessment-based approach for selection of wastewater treatment units. *National Academy Science Letters* 39(2): 103–107.

Reicherter, E., 2003. *Investigations to Indicators as a Basis for Cost Considerations in Sanitation* (in German). Oldenbourg-Industrieverl, Munich, Germany.

Rieger, L., Takács, I., Villez, K., Siegrist, H., Lessard, P., Vanrolleghem, P. and Comeau, Y., 2010. Data reconciliation for wastewater treatment plant simulation studies – planning for high-quality data and typical sources of errors. *Water Environment Research* 82: 426–433.

Roeleveld, P.J., Klapwijk, A., Eggels, P.G., Rulkens, W.H. and van Starkenburg, W., 1997. Sustainability of municipal waste water treatment. *Water Science and Technology* 35(10): 221–228.

Rosso, D. and Stenstrom, M.K., 2005. Comparative economic analysis of the impacts of mean cell retention time and denitrification on aeration systems. *Water Research* 39: 3773–3780.

Schumacher, E.F., 1989. *Small is Beautiful: A Study of Economics as if People Mattered.* Harper and Row, New York, NY, USA.

Shannon, D., Wall, B. and Flynn, D., 2014. *Focus on Urban Waste Water Treatment in 2012*: *A Report for the Year 2012*. Environmental Protection Agency, Johnstown Castle, Ireland.

Solano, M., Soriano, P. and Ciria, M., 2004. Constructed wetlands as a sustainable solution for wastewater treatment in small villages. *Biosystems Engineering* 87(1): 109–118.

Stefanakis, A.I. and Tsihrintzis, V.A., 2012. Heavy metal fate in pilot-scale sludge drying reed beds under various design and operation conditions. *Journal of Hazardous Materials* 213: 393–405.

Strauss, M. and Montanegro, A. 2002. *FS Management – Review of Practices, Problems and Initiatives*. Department of Water and Sanitation in Developing Countries (SANDEC), Duebendorf, Switzerland.

Talebizadeh, M., Belia, E. and Vanrolleghem, P.A., 2016. Influent generator for probabilistic modeling of nutrient removal wastewater treatment plants. *Environmental Modelling & Software* 77: 32–49.

Tao, X. and Chengwen, W., 2012. Energy consumption in wastewater treatment plants in China. In *World Congress on Water, Climate and Energy*. International Water Association, London, pp. 1–6.

Tewari, P.K. and Bewtra, J.K., 1982. Alpha and beta factors for domestic wastewater. *Journal (Water Pollution Control Federation)* 54: 1281–1287.

Thaler, S., 2009. *European Wastewater Benchmarking Reporting – Discussion of Current Activities and Chances of Cross-national Comparison*. European Water Association, Hennef, Germany.

Thinkstep GaBi, 2012. GaBI LCA databases. Available online: http://www.gabi-software.com/databases/gabi-databases (accessed 16 May 2018).

Tillman, A., Svingby, M. and Lundstrom, H., 1998. Life cycle assessment of municipal waste water systems. *The International Journal of Life Cycle Assessment* 3(3): 145–157.

Tillman, A.M. and Baumann, H., 2004. *The Hitch Hiker's Guide to LCA.* 1:7 Edition. Studentlitteratur, Lund, Sweden.

Torregrossa, D., Schutz, G., Cornelissen, A., Hernández-Sancho, F. and Hansen, J., 2016. Energy saving in WWTP: daily benchmarking under uncertainty and data availability limitations. *Environmental Research* 148: 330–337. Available online: http://www.sciencedirect.com/science/article/pii/S0013935116301360

Turovskiy, I.S. and Mathai, P.K., 2006. *Wastewater Sludge Processing.* John Wiley & Sons, New York, NY, USA.

US EPA, 1981. *Operation and Maintenance Costs for Municipal Wastewater Facilities*. Office of Program Operations, Washington, DC, USA.

Vidal, N., Poch, M., Martí, E. and Rodríguez-Roda, I., 2002. Evaluation of the environmental implications to include structural changes in a wastewater treatment plant. *Journal of Chemical Technology & Biotechnology* 77(11): 1206–1211.

von Sperling, M., 1996. Comparison among the most frequently used systems for wastewater treatment in developing countries. *Water Science and Technology* 33(3): 59–72.

Vymazal, J., 2005. Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment. *Ecological Engineering* 25(5): 478–490.

Vymazal, J., 2010. Constructed wetlands for wastewater treatment. *Water* 2(3): 530–549.

Wallace, S.D. and Knight, R.L., 2006. *Small Scale Constructed Wetland Treatment Systems: Feasibility, Design Criteria and O&M Requirements.* IWA Publishing, New York, NY, USA.

Wendland, A., 2005. *Operation Costs of Wastewater Treatment Plants*. Available online: https://cgi.tu-harburg.de/~awwweb/wbt/emwater/documents/lesson_c2.pdf

White, F.M. 2003, *Fluid Mechanics.* 5th edition. McGraw-Hill Book Company, Boston, MA.

Woodward, D.G., 1997. Life cycle costing – theory, information acquisition and application. *International Journal of Project Management* 15(6): 335–344.

Yang, L., Zeng, S., Chen, J., He, M. and Yang, W., 2010. Operational energy performance assessment system of municipal wastewater treatment plants. *Water Science and Technology* 62(6): 1361–1370.

Abbreviations

AAO	Anaerobic anoxic oxic
ADPf	Abiotic resource depletion (fossil)
AO	Anoxic oxic
AP	Acidification potential
BOD	Biochemical oxygen demand
CAPEX	Capital expenditure
CAS	Conventional activated sludge
cBOD	Carbonaceous biochemical oxygen demand
CCC	Concordance Correlation Coefficient
CCI	Construction cost index
CMAS	Complete mix activated sludge
CML	Institute of Environmental Sciences
COD	Chemical oxygen demand
CW	Constructed wetland
DSC	Dry solids concentration
DST	Decision support tool
EA	Extended aeration
EP	Eutrophication potential
EPA	Environmental Protection Agency
GWP	Global warming potential
HF	Horizontal flow
HLR	Hydraulic loading rate
HTP	Human toxicity potential
ICW	Integrated constructed wetland
IFAS	Integrated fixed-film activated sludge
IWA	International Water Association
KPI	Key performance indicator
KPICalc	Key Performance Indicator Calculator
LCA	Life cycle analysis
LCC	Life cycle cost
LCCA	Life cycle cost assessment
LCI	Life cycle inventory
MBBR	Moving bed biofilm reactor
MBR	Membrane bioreactor
NPV	Net present value
OD	Oxidation ditch
ODP	Ozone depletion potential
OLR	Organic loading rate
OPEX	Operational expenditure
PE	Population equivalent
PV	Present value
RBC	Rotating biological contactor
SBR	Sequence batch reactor
SPV	Single present value
SRT	Solids retention time
TF	Trickling filter
TN	Total nitrogen
UPV	Uniform present value
VF	Vertical flow
WAS	Waste activated sludge
WSP	Waste stabilisation ponds
WWTP	Wastewater treatment plant
WWTS	Wastewater treatment system

Appendix 1 Survey of Existing Irish WWTSs

Table A1.1. Licensed treatment systems as a percentage of total licensed treatment systems in Ireland

Treatment system	Percentage
Biofilter	0.63%
CAS	36.27%
EA	7.97%
ICWs	0.42%
IFAS	0.21%
MBR	0.21%
MBBR	0.21%
Pump flow bioreactors	0.21%
RBC	5.87%
SBR	13.84%
TF	2.94%
Unspecified	31.45%

Information supplied by the EPA (2014).

A1.1 Capital Expenditure Adjustment

$$C_c = \left(\frac{I_c C_t K_l}{I_t}\right) \times ER_l \qquad \text{(A1.1)}$$

where C_c is the current cost of the system, I_c is the current construction cost index (CCI); I_t is the CCI at time *t* of plant construction; C_t is the cost of construction at time *t*; K_l is the location factor: Ireland–US location factor in 2015 was 1.3 and the Ireland–Greece location factor was unavailable (assumed factor of 1);[10] and ER_l is the currency exchange rate (the euro–US dollar rate in 2015 was approximately 0.9).

Table A1.2. Overview of some life cycle analysis studies conducted in Ireland

Study title	Author(s)	Funding body/academic institute	URL
Life cycle assessment of Irish compost production and agricultural use	Eoin White	Department of the Environment, Community and Local Government	http://www.cre.ie/web/wp-content/uploads/2010/12/Compost-Life-Cycle.pdf
Development and evaluation of life cycle assessment technologies for the Irish dairy industry	Laurence Shalloo	Teagasc	https://www.teagasc.ie/media/website/publications/2016/6421_Technology_Update_LCA_Technologies_L_Shalloo.pdf
Quantification of the potential of white clover to lower greenhouse gas emissions from Irish grassland-based dairy production	James Humphreys	Department of Agriculture, Food and the Marine	https://www.agriculture.gov.ie/media/migration/research/rsfallfundedprojects/2007projects/RSF07516FinalReport.pdf
Use of life cycle assessment in Irish freshwater aquaculture systems	Ronan Cooney, Robert Walsh, Richard Fitzgerald and Eoghan Clifford	Department of Agriculture, Food and the Marine	http://www.morefish.ie/wp-content/uploads/2016/12/Environ-LCA.pdf
Life cycle assessment of electricity production in Ireland	Deidre Wolff	Dublin Institute of Technology	http://dit.ie/dublinenergylab/media/ditdublinenergylab/seminars/Deidre%20Wolff.pdf
Miscanthus production and processing in Ireland: an analysis of energy requirements and environmental impacts	Fionnuala Murphy, Ger Devlin and Kevin McDonnell	University College Dublin	http://irserver.ucd.ie/bitstream/handle/10197/5655/Environmental_and_energy_performance_of_Miscanthus_production_and_processing_systems_in_Ireland.pdf?sequence=1

10 The location factor normalises the differences in cost of construction between countries.

A1.2 Life Cycle Cost Model

The SPV method (equation A1.2) applies to one-off payments that occur infrequently sometime in the future. It is used in the present study to account for replacement parts and the system's residual value at the end of its lifetime.

$$SPV = \frac{C_o}{(1+d)^n} \tag{A1.2}$$

where C_0 is the original cost at the base year; n is the number of years from the base year; and d is the applied discount rate (nominal discount rate of 3.5% applied). Annually recurring operational costs are calculated with the UPV formula (equation A1.3).

$$UPV_{OM} = \sum A_{0,i}\left(\frac{(1+d)^n - 1}{d(1+d)^n}\right) \tag{A1.3}$$

where $A_{0,i}$ is the annual recurring cost of the operation and maintenance element i, at base year 0. In the study conducted by Rawal and Duggal (2016), recurring energy costs were treated separately from other O&M costs. This relates to the volatility in the cost of energy. In recent years, changes in the cost of energy have not aligned with CCIs; therefore, a separate discount rate for the energy UPV (UPV_E) of should be used (equation A1.4). The total LCC of each WWTS is given by equation A1.5.

$$UPV_E = A_O \frac{(1+d)^n - 1}{d(1+d)^n} \tag{A1.4}$$

$$LCC = \sum(SPV + UPV_{OM} + UPV_E) \tag{A1.5}$$

A1.3 System Analysis Loading and Discharge Limit Values

Table A1.3. Typical concentrations of wastewater pollutants

Parameter	Concentration (mg/l)	
	High	Low
BOD	350	230
TSS	400	250
TN	60	30
TP	15	6
NH_3	45	20
PO_{43-}	10	4

TP, total phosphorus, TSS, total suspended solids.

Table A1.4. Discharge limit variation (values in mg/l)

Discharge limit band	BOD	NH_3	PO_4^{3-}	TN
1	30	n/a	n/a	n/a
2	30	1	n/a	n/a
3	30	1	0.5	na/
4	30	0.5	0.5	15

n/a, not applicable.

Table A1.5. Sludge treatment options

Sludge option number	Description	Specific disposal cost (€/m³)
1	Dewatering – land spreading	60
2	No dewatering – external contractor – land spreading	75
3	Drying beds – external contractor – land spreading	60

Table A1.6. Sludge dry solids concentrations assumed for the study

Sludge type	Range of DS concentrations (%)	Assumed value (DS) (%)	Reference
Primary	2–7	4.3	Turovskiy and Mathai (2006)
Drum screen	2–7	4.3	Assumed similar removal rates to those of primary setting
SBR	2.6–5.7	4.3	Janczukowicz *et al.* (2001)
Waste activated	0.4–1.5	1.3	Turovskiy and Mathai (2006)
Attached growth	1–4	2.5	Turovskiy and Mathai (2006)
All-in-one dewatering and thickening unit		24	PWTech
Drying beds	40–70	50	Strauss and Montanegro (2002)

DS, dry solids.

Table A1.7. Sludge disposal options and specific costs

Sludge type	Disposal option	Specific costs	Source
Untreated			
	Transport to parent plant	€0.66/m^3 per km	Mooney Transport, Birr, County Offaly, sales representative, personal communication, December 2016
	External contractor[a]	€75/m^3	Enva Ireland, sales representative, personal communication, 15 November 2016
Treated			
D+S[b]	Land spreading	€60/kg	Personal communication
D[c]	Transport to parent plant	€0.66/m^3 per km	Personal communication
D	External contractor	€75/m^3	Personal communication

[a]Enva is a waste management company in Ireland that provides sludge stabilisation and disposal services.

[b]Dewatered and stabilised.

[c]Dewatered only.

Table A1.8. Aeration system parameters, reported value ranges and assumed values

Parameter	Variation/range	Assumed values	Source
Aerator system	Submerged diffuser horizontal surface (rotary type)	n/a	n/a
Diffuser types	Fine bubble diffusers and coarse bubble diffusers	n/a	n/a
Oxygen transfer efficiency	Range (kg O_2/kWh)	n/a	n/a
Surface aerator	1.5–2.1	1.8	Henze (2008), Bolles (2006)
Fine bubble diffusers	3.0–4.8	3.5	
Alpha factor (α)			
Surface aerator	0.85	Function of SRT	Rosso and Stenstrom (2005)
Fine bubble diffusers	Variable		
Beta factor (β)	0.97–0.99	0.9	Tewari and Bewtra (1982)
Fouling factor	0.4–1	0.9	Garrido-Baserba et al. (2016)
Tank depth (m)	4–6	Variable based on tank surface area to depth ratio	n/a
Tank shape	Rectangular, round	n/a	n/a
Blower efficiency	0.45–0.65	0.60	Metcalf & Eddy (2014)
Motor efficiency	0.85–0.95	0.90	Metcalf & Eddy (2014)
Temperature (°C)	Variable	10	n/a
Elevation (metres above sea level)	Variable	118	n/a

n/a, not applicable.

Table A1.9. Pumping model parameters and assumed values

Variable	Influent	Primary sludge	WAS	RAS	Nitrate recycle	TF	Source
ΔH (m)	3	7	7	3	0	Variable	
L_{pipe} (m)	8	Variable	Variable	Variable	Variable	Variable	
D_{pipe} (m)	0.1–0.15	0.1–0.15	0.1–0.15	0.1–0.15	0.1–0.15	0.1–0.15	Jones *et al.* (1989)
Minimum fluid velocity (v) (m/s)	1.83	1.83	1.83	1.83	1.83	1.83	Poloski *et al.* (2009)
Fluid density (ρ) (kg/m^3)	1010	1030	1010	1010	1010	1010	
Solids concentration (%)	0.1	4.3	1.3	0.8	0.35	0.8	Turovskiy and Mathai (2006)
Viscosity (μ) of water (N s/m^2)	1.25×10^{-3}						
Sum of the minor headloss coefficients (Σk)	12.5	9.6	9.6	8	8	12.5	White (2003), Jones *et al.* (1989)
Motor efficiency (η_m)	0.8	0.8	0.8	0.8	0.8	0.8	Jones *et al.* (1989)
Pump efficiency (η_p)	0.55	0.55	0.55	0.55	0.55	0.55	Metcalf & Eddy (2014)
Mulbarger friction factor (m_f)	n/a	1.75	n/a	n/a	n/a	n/a	Jones *et al.* (1989)

RAS, return activated sludge.

Table A1.10. Energy use assumptions for common unit processes

Unit process	Value	Details	Reference
Mechanical inlet screens (kWh/m^3)	0.01	Continuous belt type	Foladori *et al.* (2015)
Primary sedimentation tanks (kWh/m^3)	0.012	Circular tank	Foladori *et al.* (2015)
Secondary sedimentation tanks (kWh/m^3)	0.012	Circular tank	Foladori *et al.* (2015)
Thickening and dewatering (kWh/kg DS)	0.05	All-in-one unit	Amcom
Municipal energy (kWh/m^3)	0.012	Plant lighting, control and automation, administration buildings	Foladori *et al.* (2015)

DS, dry solids.

Table A1.11. Chemicals and specific costs

Chemical	Formula	Cost	Reference
Ferric chloride (37% concentration)	$FeCl_3$	€0.70/l	Personal communication, 2016, Acorn Water, Bandon, Co. Cork, Ireland
Sodium hydroxide	NaOH	€0.77/kg	Kemcore
Calcium hydroxide[a]	$Ca(OH)_2$	€0.20/kg	Index Mundi
Polymers (acrylic acid)	variable	€5/kg	Keller Schnier
Calcium hypochlorite[b]	$Ca(OCl)_2$	€1.53/kg	Kemcore
Ethanol	C_2H_6O	€0.65/l	Ng-Tech

[a]Estimated cost is based on US values adjusted from 2013 to 2017.

[b]Original price was quoted for 65% available chlorine; price presented here has been adjusted to represent 100% chlorine.

Table A1.12. Lime stabilisation dosage

Sludge type	Solids concentration (%)	$Ca(OH)_2$ dosage range (g/kg DS)	Model values (g/kg DS)
Primary	4.3	60–170	120
Secondary	1.3	210–430	300
Mixed sludge (60:40) P&S	3.8	n/a	192

Adapted from data from Metcalf & Eddy (2014).
DS, dry solids; n/a, not applicable; P&S, primary and secondary.

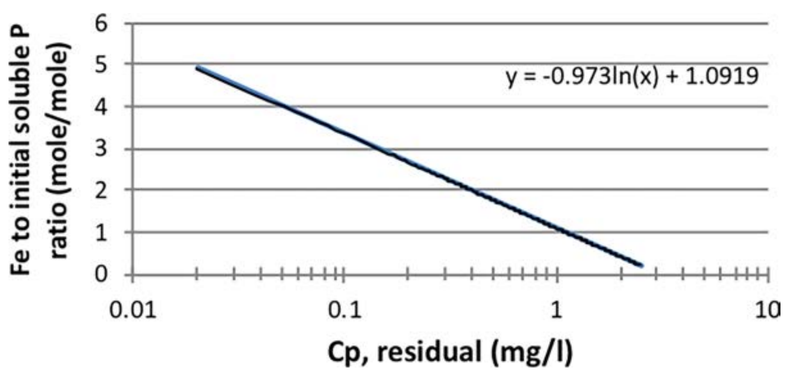


Figure A1.1. Required iron (Fe) as a function of influent phosphorus concentration. Adapted from Metcalf & Eddy, 2014.

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL
Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialú: *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

Eolas: *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

Tacaíocht: *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

Ár bhFreagrachtaí

Ceadúnú
Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:
- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha)*;
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil
- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Uisce
- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol
- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn
- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

Taighde agus Forbairt Comhshaoil
- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

Measúnacht Straitéiseach Timpeallachta
- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

Cosaint Raideolaíoch
- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Faisnéis Inrochtana agus Oideachas
- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol *(m.sh. Timpeall an Tí, léarscáileanna radóin)*.
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

Múscailt Feasachta agus Athrú Iompraíochta
- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil
Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:
- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- Oifig um Chosaint Radaíochta agus Monatóireachta Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

EPA Research Report 255

Optimal Design and Operation of Small-scale Wastewater Treatment Plants: The Irish Case

epaResearch
Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Authors: Lorna Fitzsimons, Greg McNamara, Edelle Doherty and Eoghan Clifford

Municipal wastewater treatment is resource intensive, typically using a combination of physical, biological and chemical processes to treat wastewater to designated environmental standards. Traditionally, the main priority of wastewater treatment has been driven by environmental effluent quality standards. However, it is important to note that effluent quality is not the only environmental impact of wastewater treatment; other important impacts include the use of energy and chemicals, emissions to air, and sludge management. The focus of this research was small wastewater treatment plants (population equivalent < 2000). The main objectives were to:

- develop a methodology and decision support tool (DST) to assess the life cycle costs and environmental impact of wastewater treatment plants for given site-specific characteristics;
- develop software tools and methodologies to assist stakeholders to benchmark to better manage Irish wastewater treatment plants.

Identifying pressures

Wastewater treatment plant managers are tasked with achieving emission limits values while simultaneously improving resource efficiency. Treatment plants vary considerably in terms of their emission limits values, scale, loading, technology and sludge management inter alia.

Benchmarking and life cycle cost assessment are two key approaches to (1) compare and improve the performance of wastewater treatment plants and (2) estimate the life cycle costs of a wastewater treatment system in given site-specific conditions. However, this research identified pressures in the adaptation and implementation of these approaches, for example limited and non-standardised data availability, and the need for effective sampling methodologies to accurately compare plant performance.

Informing policy

The Urban Waste Water Treatment Directive (91/271/EEC) requires that wastewater is treated to designated standards before being discharged to the environment. Required effluent quality is largely driven by the sensitivity of the receiving water and the scale of the treatment plant. For example, nutrient removal is required in sensitive areas that are subject to eutrophication. Nutrient removal, as well as the predicted requirement to remove emerging contaminants of concern, is expected to increase wastewater treatment plant resource consumption. As a result, there are inherent trade-offs between the positive environmental benefits of higher quality wastewater and the negative environmental impacts associated with additional energy and chemical requirements to achieve those standards. Using life cycle assessment (LCA) this research demonstrates that emission limits values, technology and operational choices have important consequences for environmental performance and therefore this should also be an important consideration for policymakers, as well as effluent quality.

Developing solutions

The research team has developed benchmarking and life cycle cost/LCA methodologies and software tools (KPICalc and DST) that holistically consider both the economic and the environmental costs of wastewater treatment from a site-specific perspective (emission limits values, scale, loading, technology, sludge management). A DST was developed, which, given accurate local cost data, can be used to compare various wastewater treatment technologies under different user-defined operating scenarios, in terms of economic and environmental impact.

This research also developed software tools to assist key stakeholders to benchmark the performance of wastewater treatment plants. The tools developed facilitate the identification of suitable wastewater treatment plants for comparison purposes and appropriate sampling strategies.

EPA Research: McCumiskey House, Richiew, Clonskeagh, Dublin 14.

Phone: 01 268 0100
Twitter: @EPAResearchNews
Email: research@epa.ie

www.epa.ie

EPA Research Webpages
www.epa.ie/researchandeducation/research/