Experimental Meso-Scale Integrated Constructed Wetlands for the Treatment of Piggery Wastewater

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Declaration.

I, Caolan Harrington, declare that the work included in this manuscript has been composed by myself. The research that was conducted in this project was part of a research project entitled “Energy generation option for pig manure and sustainable disposal of residue” as part of the Irish Department of Agriculture Food and Fisheries’ (DAFF) Research Stimulus Fund Programme (RSFP) under the National Development Plan 2007-2013. The Stimulus Research group was operating a project which encompassed experimental anaerobic digester design, composting of solids, woodchip filters, pyrolysis, constructed wetlands and microbial analysis of all techniques. The material presented in this manuscript comprised the entirety of the constructed wetlands phase of the research project. No additional persons worked on the meso-scale project other than the author.

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The work herein has not been submitted for any other degree or professional qualification.

Signed ______________________ Date: 11-October-2012

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Abstract 6

1. Introduction and Agricultural Practice 8

1.1 International Design Guidelines: Global Scenario 10

1.1.1 American guidelines 10

1.1.2 Other guidelines 17

1.1.3 Recent innovations 20

1.2 Operations 23

1.2.1 Loading and flow rates 24

1.2.2 Water depth 26

1.2.3 Pre-treatment of wastewater 27

1.2.4 Sizing of wetlands 29

1.3 Macrophytes and Rural Biodiversity 30

1.3.1 Macrophyte types and characteristics 30

1.3.2 Toxicity tolerance thresholds 31

1.4 Nutrients 33

1.4.1 Nutrient transformation processes 33

1.4.2 Phosphorus 36

1.5 Pathogens, Odour and Human Health 39

1.6 Scope of this project 41

2. Materials and method 47

2.1 Site location and description 47

2.2 Basis for the systems and their design 48

2.3 Construction 52

2.3.1 Construction methods 52

2.3.2 Homogeneity of systems and the key variables 54

2.4 Macrophytes 55

2.5 Startup 57

2.6 Operational methods 62

2.7 Sampling methods 64

2.8 Sample analysis 65

2.9 Additional sampling topics 68

2.10 Problems found and solutions 70

2.11 Overall status. 74

3. Results 75
3.1 Initial data 75
3.2 Standard treatment 77
3.2 Recycling 78
3.2 High Nutrient Loading 79
3.2 High Flow Rate 81
3.3 Statistics: 92
3.3.1 Cell1 outflow analysis 94
3.3.2 Cell 4 outflow analysis 102
3.4 Treatment differences 110
3.4.1 Cell 1 outflow treatment differences 111
3.4.2 Cell 4 outflow treatment differences: 112
4. Discussion 113
4.1 Initial removal of nutrients 116
4.2 Ammonia reductions 117
4.3 MRP reductions 118
4.4 Mass nitrogen removal 119
4.5 Pathogen reductions 121
4.5 Stability of the systems 122
4.6 Simple mechanics and reliability 123
4.7 Comparative performance 124
4.8 Costs and Land-use 128
4.9 Shortcomings of the project 133
4.10 Applications 137
4.10.1 Research application 137
5. Conclusion 139
5.1 Key operations of the meso-scale ICW 139
5.2 The use of ICW as a viable option for piggeries 140
5.3 The use of the meso-scale design as a research tool 140
References 144

Abstract

Since the 1950s, Constructed Wetlands (CW), have seen an ever-increasing rise in their popularity as a viable and alternative method of wastewater treatment. From small beginnings to a surge of guidance documentation in the latter part of the 20th century, they have undergone many revisions and studies in their design, implementation and operation. Several significant American guidance documents for the design of constructed wetlands were published in the late 1980's. These studies have led to great variations in design and performance of constructed wetlands for the treatment of high-strength ammonia-nitrogen (370mg /L^{-1} - 230mg L^{-1}, 8-25 kg ha^{-1} day^{-1}) polluted waters such as swine wastewater (also called pig slurry in Europe) dependant on their operational mode as well as their location and regional climate. Considering that treatment performances were often unacceptably low, more recent designs have focused on variables such as loading rate, hydraulic retention time, pre-treatment and recirculation of pre-treated water. Furthermore, a wide range of macrophyte species have been studied with regard to their tolerance to nutrient levels, uptake rates and climatic tolerance. The nitrogen cycle plays a vital role particularly in swine wastewater management and as such the promotion of nitrification and denitrification has been researched. However, ammonia is toxic to commonly used wetland plants. Therefore, alternative plants such as cash crops have also been examined, because they allow for constructed wetlands to have potentially additional benefits such as food production, revenue increase and employment generation.

Swine wastewater in Ireland is currently a significant issue in regards to EU Directives directly relating to water quality. This research study was performed

to examine the potential application of a constructed wetland for the effective treatment of swine wastewater. A highly-replicated system, based upon the Integrated Constructed Wetland approach, pioneered in Ireland was constructed, operated and sampled for an 18-month period in the South of Ireland. The system received separated liquid from an anaerobic digestor unit and this liquid was fed into the system for 18 months. Weekly sampling and analysis showed that at low influent concentrations (up to 200mg NH_4/l), the systems were capable of removing nitrogen species to background levels and producing a discharge that met some of Irelands provisional drinking water standards.

The effectiveness of these meso-scale systems highlight the potential application of the Integrated Constructed Wetland approach for the treatment of piggery wastewaters. In an industry that is heavily reliant upon other farm/land owners taking excess wastewater as a form of fertilizer, coupled with strict landspreading application limits, increasing fuel costs and EU water quality directives, they could provide a financially and environmentally beneficial approach to a robust total nitrogen management scheme.

Keywords: design guidelines; macrophytes; water quality; nitrogen removal; phosphorus retention

1. **Introduction and Agricultural Practice**

Over the last 60 years, constructed wetlands have become widespread as an alternative and cost-effective method of treating wastewaters including agriculture-related wastewater streams (Vymazal, 2010; Harrington *et al.*, 2005; Kivaisi, 2001; Scholz and Lee, 2005; Sievers, 1997, Stone *et al.*, 2002). As opposed to traditional mechanical and chemical treatment techniques such as sludge treatment plants or trickling filters, constructed wetlands implement only processes that occur naturally. The effective treatment of swine wastewaters (also called pig slurry in Europe) has been studied in greater depth over the past two decades with more and more processes being examined and scrutinised.

A substantial body of research originated in the USA, where government agency guidelines were published for the construction and operation of constructed wetlands for various applications, but predominantly for domestic wastewater and storm water treatment (USEPA 1988, 2000). However, with respect to the complex nature of diffuse agricultural pollution, these guidelines were found to be inadequate (Sievers, 1997; Stone *et al.*, 2002, 2004) and resulted in mediocre overall treatment results due to the absence of a pre-treatment stage promoting sedimentation and/or nitrification. Livestock waste and wastewaters are considered as being particularly difficult to treat through the use of constructed wetlands due to their generally higher concentrations of pollutants including suspended solids, nutrients and bacteria (He *et al.*, 2006). Moreover, an increasing number of livestock on farms and high production rates result in very high volumes of wastewater that prove to be a challenge to constructed treatment wetlands.

In the European Union, the Nitrates Directive 91/676/EEC (EEC, 1991) has put presure on piggeries due to the restrictions placed on land-spreading of corresponding wastewaters at certain times of the year. Currently in Ireland, this directive places a limit of 170kg of organic nitrogen per hectare per annum on all farmland onto which slurries and wastewaters can be spread (SI378, 2006). In several regions in Ireland, organic nitrogen levels are sufficiently high that additional application of slurries is restricted and farmers offset their slurry production with the selling of it as fertilizer to other landowners. In cases where the sale of slurry is not practical, storage is limited, or disposal is prohibitively expensive, some look toward the use of anaerobic digestion as a potential use of slurries (Nolan, *et al*., 2012). Considering that suitable farm land for spreading is limited and other land owners are often unable to spread piggery waste as fertiliser, expensive and high-maintenance equipment is often purchased that can put great financial burdens on piggeries and farmers. The holistic approach of using constructed wetlands for the treatment of wastewater is therefore rapidly becoming a far more appealing approach to many farmers. However, constructed wetland acceptance will be hindered without greater understanding of the processes involved and guidance to their construction, operation and maintenance.

The initially widely-used guidelines from the USEPA (1988, 2000) were sufficient for certain types of wastewaters. Long and narrow systems yielded often respectable results despite high flow velocity measurements. This basic design has moved on to more sophisticated wetland system design adopted to cold, temperate and tropical climate (Kantawanichkul, 2003; Puustinen, 2005). Different designs include vertical-flow, horizontal surface flow and sub-surface flow systems (Scholz, 2006).

The basis for the use of constructed wetlands for wastewater treatment is well established and its ‘fine-tuning’ for the use of intense agricultural wastes is on-going.

However, some systems such as integrated constructed wetlands have produced exceptionally positive treatment results for farmyard runoff and domestic wastewater (Scholz *et al.*, 2007), and are therefore being considered also for swine wastewater treatment.

This section covers the current published knowledge and understanding with respect to the treatment of wastewater high in ammonia-nitrogen (370mg NH_3/l, Poach *et al*, 2002; 230mg NH_3/l, Vanotti and Hunt, 2001; 16-32 kg ha/d, Reddy *et al*, 2001; 8-25 kg ha/d, Stone, Hunt and Szogi, 2000) with constructed wetlands. Swine wastewaters are the primary focus, but with the multi-faceted use of constructed and treatment wetlands for various wastewaters, some overlap between municipal, industrial and agricultural is given. The review will focus on swine wastewater treatment representing a 'worst case scenario' for wetland designers. The reasoning for further research needs incorporating rather multi-disciplinary and holistic views is also outlined.

1.1 International Design Guidelines: Global Scenario

1.1.1 American guidelines

Since the early 1980s, a wide range of constructed wetland publications originating from the USA were promoted. These include guidance manuals, case studies, bibliographies, assessments of technologies involved (both free surface water wetlands and sub-surface flow wetlands) and the availability of a description and performance database; the so called North America Treatment Wetland Database, which is based on constructed wetlands currently in use (Knight *et al*., 2000).

The US Environmental Protection Agency (USEPA) released a manual in 1988 for the design of constructed wetlands for treatment of municipal wastewaters (USEPA, 1988). The design parameters presented in the manual are based on a large number of empirical formulae and led to what is known as the "rational method" (Sievers, 1997), which predicts wetland area based upon desired organic (five day biochemical oxygen demand) removal. These designs, while stating that construction would be dependent upon the topography of the site, are usually long and linear systems that comprised small number of cells and often had deep sections of water.

Table 1. Overview of design codes for constructed wetlands

Country	Reference	Year	Recommended area	Aspect ratio	Liners
America	USEPA (2000)	2000	$ALR = Q\ C_0/A_w \times 4$	3:1 (up to 5:1)	When needed
Scotland / Northern Ireland	SEPA (2008)	2008	Twice the capture area	>2.2:1	When needed
Finland	SYKE (2005)	2005	x% of catchment area	Dependent upon topology	No
Ireland	DoEHLG	2010	Farmyard: (2 x interception area*) x 1.25**	Dependent upon topology	No
			Domestic: (PE x 20 to 40***) x 1.25**	Dependent upon topology	No

A_w : Wetland area (ha); ALR : Average loading rates; Q : incoming flow rate (m^3/d); C_0 : influent concentration, in (mg/l); * Full farmyard, inclusive of roof areas and any other areas requiring run-off capture; ** Supporting infrastructure, area taken up by embankments and associated access; *** 20m^2/PE for wastewater, 40m^2/PE for wastewater that includes stormwater.

The specific processes taking place in constructed wetlands are similar to those that occur in conventional treatment systems including treatment methods designed to promote denitrification. The use of emergent vegetation is crucial in both sub-surface and free surface water systems. The emergent vegetation controls water flow and facilitates the main processes of treatment through their rhizosphere where most treatment processes take place.

The USEPA (1988) design manual recommends the use of constructed wetlands for the treatment of acid mine drainage and storm water. It further suggested that they may enhance existing predominantly natural wetlands. The manual acknowledged the considerable costs of more 'traditional' treatment systems, such as sand filters, mechanical treatment, additions of flocculants, anaerobic digestion and settling tanks, and emphasized that constructed wetlands are a cost effective alternative, which are based on natural processes that occur in natural wetlands. The construction of wetlands in areas where they did not previously exist highlights that there is less regulation with regard to constructed wetlands than in comparison to natural wetlands. These and similar aspects about the constructed wetlands ease of implementation and construction have been the driving factors behind the development of constructed wetlands for wastewater treatment since the late 1970s. Moreover, their performance, especially when compared to industrial methods, such as sand filters, mechanical treatment, additions of flocculants, anaerobic digestion, is a crucial factor promoting their increase of acceptance (Scholz, 2006).

The early design manual (USEPA, 1988) was followed up by a series of guidance documents and manuals in the USA and elsewhere, such as Scotland & Northern Ireland and Italy (Carty *et al*., 2008; Conte *et al*., 2001; USDA, 1991; USEPA, 2000). Similar guidelines were released by the United States Department of Agriculture under the Natural Resource Conservation Service (USDA, 1991). This handbook recommends constructed wetlands for the treatment of agricultural and domestic wastewater, mine drainage and storm water. Similar to the USEPA (1988) manual, it also relies on the constructed wetland design based predominantly on biochemical oxygen demand concentrations, but expanded the design parameters by including ammonia-nitrogen as the new key design factor and recommended the implementation of a minimum twelve–day residence time for the influent to be treated.

Both manuals are similar but have led to mixed results. Some reports have shown that nutrient concentrations were significantly higher than expected (Sievers, 1997) when using the USEPA (1988) guideline. In comparison, others have shown that results are directly in line with the expected effluent quality using the USDA (1991) guideline (Stone *et al*., 2002).

The manual "Constructed Wetland Treatment of Municipal Wastewaters" (USEPA, 2000) is recommended for the design of constructed wetlands as a functional part of a wastewater management strategy, and not as a guidance manual for a "specific regulatory program". It therefore gives in-depth guidance for both free surface flow wetlands and vegetated submerged beds.

The USEPA (2000) manual describes constructed wetlands and the associated terminology, applications and apparent misconceptions in a manner that is easy to understand. It allows for a wide array of people to comprehend the design, application and implementation of constructed wetlands. However, its attempt to clarify certain aspects of constructed wetland design is confusing. On one hand, it explains that constructed wetlands are well defined by equations based upon a wide range of literature summarizing good research, but states on the other hand that they cannot be designed purely by such equations. The document then gives example designs using these equations for the design of constructed wetlands for municipal wastewater treatment without outlining other design criteria in detail.

What has been consistent in the implementation of constructed wetlands is the focus on phosphorous treatment. It had been shown on several occasions that initial treatment was quite effective, but that after a relatively short period of time the reduction levels reduced considerably (Scholz, 2006; Sievers, 1997; Stone *et al*., 2002). As phosphorous is deemed very important with regard to water quality, several studies recommend constructed wetlands only as a method of pre-treatment when based upon the USEPA (1988) and USDA (1991) manuals.

The Gulf of Mexico program that was established in 1988 was an inter-agency effort in the USA to improve both ecological and economic viability of the Gulf of Mexico. As part of this programme, the Nutrient Enrichment committee recognised the potential of “on-farm treatment wetlands” (Knight *et al*, 2000) for the reduction of pollutants that were entering the Gulf of Mexico watershed. As a result of this, in 1995 the program funded a project which assessed the use and effectiveness of treatment wetlands for wastewater management and made the gathered data available to the public. This data has been used in conjunction with the North American Treatment Wetland Database to compare a wide range of surveyed and catalogued treatment wetlands in the USA receiving livestock wastewaters. The USEPA online database, North American Treatment Wetland Database (NADB) catalogued 206 natural and constructed wetlands between 1991 and 1993. An updated version included wetland data on the wetlands catalogued, including locations, populations, designs, costs, operating data for water quality and habitats. The NADB was modified to develop the Livestock Wastewater Treatment Database (LWDB). This was in order to accommodate the characteristics of livestock (dairy, cattle, swine, poultry and aquaculture) wastewater treatment wetlands. These two databases were combined in the updated version of the NADB.

These databases showed that the area for most of the catalogued wetlands treating livestock wastewater were of a very small size, averaging at 0.6 ha. The majority of the wetlands treating swine, poultry and dairy wastewater were less than 0.1 ha. These were predominantly research scale systems and not large-scale operational wetlands (Knight *et al*, 2000; Sievers, 1997; Stone *et al*., 2002). Sievers (1997) implemented the "Rational Method" laid out by the USEPA (1988) guidelines for 2 different wetland designs (sub-surface flow and free water surface flow). The Rational Method predicts the area required for a wetland based upon a desired BOD_5 removal and calculated from empirical formulae. The results from these cells were highly variable; most likely due to seasonal changes. The inability of the constructed wetland to consistently achieve a biochemical oxygen demand concentration of 30 mg/l or less was presumed to be a result of the wetlands maintaining anaerobic conditions. Over a two year study period, no nitrate was detected in the constructed wetland system, thus suggesting insignificant nitrification. Phosphorous removal was also very low. The primary removal method was adsorption on soil with uptake by plants secondary (Tchobanoglous, 1987). Despite of low overall removal rates (15% - 27% removal of phosphorus), the cells receiving wastewater from the primary lagoon provided comparatively better removal rates than the system receiving influent from the secondary lagoon, most likely due to a longer retention time (7 and 9 days compared to 3 and 4 days respectively).

Stone *et al*. (2002) used the Natural Resources Conservation Service presumptive design method. This involved four wetland cells (i.e. two cells connected in series). These cells had a length to width ratio of 9:1, and were both planted with the macrophyte/helophyte bulrushes/cattails (Typha). Over a six year period, the macrophyte was examined while receiving effluent from an anaerobic lagoon, which was mixed with fresh water prior to entering the wetland.

The wetlands were found to be relatively effective at the removal of nitrogen with mean total nitrogen and ammonia-nitrogen reductions of approximately 85% each over the duration of the study period. At lower rates, the total phosphorous removal rate was approximately 88%. Total phosphorus removal was reduced to 25% and 38% for the two systems, if operated under higher loading rates. Stone *et al*. (2002) suggested that pre-treatment would be required for the effective removal of phosphorous when high loading rates were applied. The constructed wetland sizing that was applied to these systems was dependent upon the loading rates and the concentrations of the wastewater being treated. This was found to result in a constructed wetland, which was only slightly bigger (approximately 5%) than a similar system designed according to Natural Resources Conservation Service guidelines (see above).

The American guidance manuals (USDA, 1991; USEPA, 1988, 2000) have been implemented for use in a wide range of applications with regard to various wastewaters. However, their usefulness regarding the treatment of swine wastewater was seen to be limited. Most constructed wetland designs were seen to lack the treatment capability to reach set standards for swine wastewater without pre-treatment (Stone *et al*., 2002; Sievers *et al*., 1997). The manuals, while trying to be robust and highly descriptive, are over-burdened with technical data and specific empirical formulae that are supposed to be applied to a treatment method that is extremely complex in both the reactions that are taking place in the constructed wetland as well as the external influences such as climatic conditions. These circumstances are stated in the more recent USEPA (2000) guidance manual for municipal wastewaters, and described as a misconception.

Changes of the retention time impact on the effectiveness of constructed treatment wetlands. The use of the Natural Resources Conservation Service method requires a minimum retention time of twelve days (Stone *et al.*, 2002). Sievers (1997) noted improved performances in cells with longer retention times. This observation is taken into account in the more recent USEPA (2000) guidance manual. Earlier versions recommended between six and seven days retention time. The American guidelines (USEPA, 2000) for municipal water treatment are extensive and provide a large amount of information on all aspects that are involved in wetland design. They are recommended for relatively small communities (treating less than 3,800m^3/d), which can sustain the “expertise required to maintain them” (USEPA, 2000). These guidelines provide information to both technical (engineers, regulators and planners) and non-technical readers (decision makers and stakeholders). However, there are no specific guidelines with regard to the implementation of constructed wetlands used for the treatment of agricultural wastewater. Municipal guidelines (USEPA, 1988, 1991, 2000) have been implemented by researchers and operators to examine their potential for use in swine wastewater treatment. Where the designs were based upon desired BOD_5 removal, the Rational Method for the sizing of wetlands has been shown to be inadequate in reaching the desired performance (Sievers, 1997). Other reviews of constructed wetlands based on early USEPA/USDA guidelines have shown variable BOD_5 as well as nutrient (e.g. phosphorus) removal rates (17% - 97% and 54% - 93% respectively (Cronk, 1996).

1.1.2 Other guidelines

Similar to what has been published in the USA, many other countries have their own guidance manuals for the use of constructed wetlands for the treatment of various wastewaters. Early guidelines were adapted for semi-arid (Australia), boreal (Finland), tropical (Thailand), Mediterranean (Italy) and temperate (Ireland, England and Scotland) climates, as discussed elsewhere (Carty *et al.*, 2008; Conte *et al.*, 2001; Puustinen and Jormola, 2005).

The guidance manuals are predominantly used for the design and construction of wetlands for the treatment of municipal wastewater as well as storm water (Stone *et al.*, 2002). Some of these guidelines are relatively new (Conte *et al.*, 2001; Puustinen and Jormola, 2005) in comparison to some that have been researched and implemented in other regions, namely Australia. A large proportion of the material that is used in the USA is based on data from Australia (USEPA, 1988, 2000). Australia has been using constructed wetlands for several decades for the treatment of storm water in urban areas. Indeed, similar to the guidance material in the USA, other manuals and documents also primarily recommend constructed wetlands for the treatment of (pre-treated) municipal wastewaters and storm water (Conte *et al.*, 2001).

Several countries with areas characterized by a tropical climate such as China, Thailand, Malaysia, Brazil and Australia have implemented constructed wetlands for wastewater treatment (He, 2006; Kantawanichkul, 2003; Li *et al.*, 2008; Sezerino *et al*, 2003). The functionality of constructed wetlands and their efficiency and size is influenced by the significantly higher temperature of their surroundings. Tropical constructed wetlands, like their temperate counterparts in Europe and the US, are predominantly used for storm water and wastewater management (Kantawanichkul and Somprasert, 2005), but are also being implemented for the treatment of agricultural waste and wastewater. The outline designs that are in use vary very little from those in most other regions, and the main differences are in wetland system operation. Specific guidance manuals have not been developed, and notes refer to the same design types as those used elsewhere. However, a combination of various types of constructed wetlands has been used in some case studies where space was limited (Kantawanichkul, 2003, 2005).

Due to the high robustness of constructed wetlands and their wide treatment applications, they can even be found in harsh Boreal climates that have extreme weather conditions. For example, the use of constructed wetlands in Finland for agricultural wastewater treatment is relatively new in comparison to other countries (Puustinen and Jormola, 2005). Finnish designers rely heavily on optimizing the hydraulic efficiency, preventing flows that are linear, and preferring diffuse flows instead.

Constructed wetlands, for example, in Finland are designed and built with a broader spectrum of use than just a single purpose such as municipal or agricultural wastewater treatment (Puustinen and Jormola, 2005). They are designed for floodwater mitigation, wastewater treatment, habitat creation, biodiversity support and aesthetic functionality (Puustinen and Jormola, 2005). The primary Finnish research site is the Hovi constructed wetland in southern Finland (Puustinen and Jormola 2005). In the material documenting this wetland, it states that the size of the constructed wetland with regard to the catchment area is critical. Examples stating that for >20% loading reductions, the water catchment to wetland area ration would have to be more than 2% (Koskiaho and Puustinen, 2005). This data is referenced against data from constructed wetlands in the USA and compared with data from Finland (Puustinen and Koskiaho, 2003).

The high number of studies performed on this single wetland, which was established in 1998, has resulted in two manuals (Puustinen and Jormola 2005) provided by the Finnish Environment Institute. One manual gives an overview of the implementation of constructed wetlands, their role and general information, and the other one provides specific design guidelines.

In the 1990s, there were several design guidelines that were accepted as certified technologies (Vymazal, 2010) in Austria, Denmark and Australia. Section 1.1.3 covers recent innovations in Ireland and Scotland where more recent design approaches have been implemented into fully governmentally approved guidance documents/guidelines.

1.1.3 Recent innovations

The application of constructed wetlands for the treatment of agricultural waste and wastewater is not a new concept but is one that has been refined since the construction of wetlands became viable for this specific application (Dunne *et al.*, 2005; Hunt, Matheny and Stone, 2004; Hunt *et al.*., 2002; Knight *et al.*, 2000, Sievers, 1997). There have been several recent innovations regarding the design, operation and management of constructed wetlands. For example, different types of vertical-flow wetlands are in use (He *et al.*, 2006; Kantawanichkul, Neamkam and Shutes, 2001; Molle, Prost-Boucle and Lienard, 2008; Sezerino *et al.*, 2003). Some of these systems are common for swine wastewater treatment due to their increased capacity to remove organic matter and ammonia-nitrogen (He *et al.*, 2006). Different methods of introducing wastewater into constructed wetlands (e.g. continuous, batch and “tidal” flow operation) and “anti-sized reed bed systems”, where in contrast to normal vertical-flow systems with finer material at the top of the system, the coarser material is located at the top instead, have been successfully operated (Lee, Scholz and Horn, 2006; Scholz, 2006; Scholz and Xu, 2001, Zhao, Sun and Allen, 2004). Zhao *et al.* (2004) examined the improved capability of such systems to deal with the clogging of the bedding material, as well as avoiding the possible detrimental effects of clogging on functionality and sustainability. The study had found very little difference in the actual removal capacity of nutrients in the systems, but the accumulation of material in the setups was significantly slower in the anti-sized systems. In situations where vertical-flow wetlands are used, this would lead to the improved sustainability of the constructed wetland, especially in more remote areas where access to skilled personnel would be more difficult (USEPA, 2000).

Another, more encompassing approach has been undertaken in Ireland with regard to the overall design, construction and management of wetlands. The integrated constructed wetland concept has been developed for the design of wetlands for agricultural wastewater, including swine wastewater, interception and the design takes into account social, economic and environmental, variables before construction (Harrington *et al.*, 2005). The integrated constructed wetland approach, such as the Anne Valley Project (Harrington and Ryder, 2002) also integrates wetlands into the surrounding landscape to give a more natural appearance of the overall structure (Scholz *et al.*, 2007).

Carty *et al.* (2008) have published the scientific justification for the Farm Constructed Wetland Design Manual for Scotland and Northern Ireland. This document addresses an international audience interested in applying wetland systems in the wider agricultural context. Farm constructed wetlands combine farm wastewater treatment with landscape and biodiversity enhancements, and are a specific application and class of integrated constructed wetlands, which have wider applications in the treatment of other wastewater types such as domestic sewage (Scholz *et al.*, 2007). Carty *et al.* (2008) discuss universal design, construction, planting, maintenance and operation issues relevant specifically for farm constructed wetlands including wetland systems treating piggery wastewater in a temperate climate, but highlights also catchment-specific requirements to protect the environment.

Furthermore, the design suggestions by Healy, Rodgers and Mulqueen (2007) complement the guidelines proposed by Scholz *et al.* (2007) and Carty *et al.* (2008). Healy *et al.* (2007) discuss the performance and design criteria of constructed wetlands for the treatment of domestic and agricultural wastewater, and sand filters for the treatment of domestic wastewater. It also proposes sand filtration as an alternative treatment mechanism for agricultural wastewater and suggests corresponding design guidelines.

Chen *et al.* (2008) examined modified free water surface flow constructed wetlands to polish treated swine wastewater. The assessed treatment process (i.e. conventional three-stage treatment scheme followed by a modified free water surface wetland (with or without plants) with a two-day hydraulic retention time) was a promising option to meet Chinese swine wastewater discharge limitations (COD: 600 mg/l, BOD: 80 mg/l, SS: 150 mg/l).

Most recently, Zhang *et al.* (2009) proposed the application of self-organizing map models as prediction tools for the performance of wetland-based agroecosystems for the treatment of agricultural wastewater to protect receiving watercourses. By utilizing the self-organizing map model, the time-consuming to measure expensive biochemical oxygen demand outflow concentrations were predicted well by other inexpensive variables, which were quicker and easier to measure. This novel approach allows for the real time control of the outflow water quality of wetland systems and potentially also of other treatment system applications.

In the last 5 years, two distinctive reviews have been performed on CW and their development since their inception have been published. One takes an overview approach to CW as a whole (Vymazal, 2010) which looks at the changes that have occurred in CW design and implementation as well as their uptake by governmental bodies on a global scale. Babatunde *et al*., (2008) meanwhile specifically looks at the development of CW, focussing on Ireland. With the emphasis of this research project being that of Ireland, it is an accurate overview of the current state of affairs in the country with regard to CW. Some distinct differences can be seen however, primarily that on the topic of phosphorous removal. Vymazal (2010) states that most types of CW are not effective at the removal of P without that addition of light-weight aggregates to help to increase P removal, whereas the Babatunde (2008) review shows that several systems have shown to be able to achieve high P removal rates without the addition of any aggregates or similar materials needing to be added. Phosphorus removal is covered in more detail in section 1.4.

1.2 Operations

There are several key design parameters that CW can be designed around, or that a certain system will focus upon depending on their role. In the U.S, there was a heavy focus on using BOD and organic loading as the principle parameter when they were being compiled as part of the USEPA (1982) cataloguing. Internationally, there is more emphasis on nutrient parameters such as ammonia-nitrogen in Europe, where there are heavy restrictions on the amount of organic-nitrogen that can be spread onto farmland. In terms of more industrial systems, there can be a focus on the removal of heavy metals and toxics rather than more common nutrients and as a result, they modify their designs to incorporate media in to the beds of the wetlands which may help to attenuate additional materials.

For the purposes of this research project, the focus was on parameters that are common to the vast majority of wetland designs from around the world, regardless of their primary influent type, CW role or CW design. These parameters are fundamental to most designs and are that which can be easily adjusted and examined without a great cost.

1.2.1 Loading and flow rates

The operation of constructed wetlands (Knight *et al*, 2000; Stone *et al*, 2002; Sievers, 1997; Poach *et al*, 2003; Scholz *et al*, 2007; Harrington and Ryder, 2002; Carty *et al*, 2008; Kantawanichkul *et al*, 2001) is relatively uniform, regardless of the flow type (e.g. surface flow, sub-surface flow or vertical-flow). The influent enters the wetland system and flows through one or a series of wetland cells, which are usually heavily vegetated. Where the differences lie, are the various aspects with regard to specific designs, such as length to width ratios, loading rates, water depth and shape. This can be seen through the literature from around the world in the differences in the designs implemented (USEPA, 2000; Sievers, 1997; Puustinen and Jormola, 2005; Carty *et al*, 2008; Harrington and Ryder, 2002; Scholz *et al*, 2007)

The design of constructed wetlands is therefore of great importance to ensure that they perform well, and are suited to the location and to the treatment role that they are supposed to fulfil. Comprehensive research into the most appropriate loading rates, input concentrations and flow rates has been undertaken (Kantawanichkul *et al.*, 2001; Knight *et al.*, 2000; Lee *et al.*, 2004; Szogi and Hunt, 2001).

The North American Treatment Wetland Database (Knight *et al*, 1993, 2000) contains design data for over 400 constructed wetlands, both study- and full-scale systems. A study sponsored by the Gulf of Mexico Program, established a similar database, the Livestock Wastewater Treatment Database (Knight *et al.*, 2000). The initial North American Treatment Wetland Database stores information on 206 natural and constructed treatment wetlands, whereas the Livestock Wastewater Treatment Database stores information specifically for livestock-related wastewaters. These two databases were combined due to their similarity in design to form the North American Data Base version 2.0 (Knight *et al.*, 2000). This development is important because it gives access to an in-depth catalogue of data allowing for the examination of wetland with various designs, scaling, influent/effluent types, population coverage, livestock type and quantity of influent/effluent. This allows for preliminary examination of data before any construction is done allowing for a more robust design and potentially the correction of shortcomings in any such treatment wetlands catalogued. Access to such data allows for the designer to utilize data that has been compiled in advance of any work being done and is comparable against other systems already in use. This saves on paper and field studies needing to be done to gather the same information.

The loading and flow rates in constructed wetlands vary greatly. The Livestock Wastewater Treatment Database recorded data from 68 sites in North America. The mean hydraulic loading rate was 4.7 cm/d. Mean system flows of 10 m^3/d^{-1} have been calculated. Most of the systems used for livestock wastewater treatment are of small size with a mean area of only 0.6 ha. Constructed wetlands treating swine wastewater are slightly larger with mean areas of 1 ha. Knight *et al.* (2000) concluded that concentration removal rates were a function of the inlet concentration and the hydraulic loading rates.

The USEPA (1988) guidelines recommended loading rates of 112 kg BOD_5/ha/d and flow rates of 200 m^3/ha/d. The Livestock Wastewater Treatment

Database stated mean flow rate of 10 m^3/ha/d is significantly lower than this, but the majority of the sites examined in the livestock database were research–based systems of relatively small size.

The loading rates that are used in constructed wetlands vary greatly with respect to their intended use. In dairy wastewater treatment, designers have applied very light loading rates (3.6 g/m^2/d; Dunne *et al.*, 2005), heavy rates (Lee *et al.*, 2004; USEPA, 1988), tidal flow-governed rates (Zhao *et al.*, 2004) and changing rates (Lee *et al.*, 2004). Several studies in the tropics with high hydraulic loading rates in constructed wetlands treating swine wastewater have shown that removal rates decrease with significant increases in loading rates (Kantawanichkul and Somprasert, 2005; Kantawanichkul *et al.*, 2003).

1.2.2 Water depth

The water depths in constructed wetlands are highly variable. The USEPA (1988) guidelines recommend that the variable depth should be part of the design equations that are used to determine the sizing of the corresponding constructed wetland cells. Wetlands range from shallow systems with a depth of approximately 0.25 m (Harrington and Ryder, 2002; Harrington *et al.*, 2005) to deep one of 1.2 m (USEPA, 1988). A technical report by the USEPA (1999) recorded water depths between 0.1 and 1.5 m. The depth of water is often the key parameter to nitrogen control in wastewater treatment. Shallow water depths are associated with the highest ammonia-nitrogen diffusion and nitrogen losses (Szogi and Hunt, 2001).

The water depth also has an important impact on the growth of macrophytes that are used in constructed wetland systems. Depending on the species that are planted, greater water depth can inhibit macrophyte growth and colonisation of the wetland cells. Some genera of macrophytes are capable of coping with deep water levels (Clarke and Baldwin, 2002; USEPA, 1988), but others are not, and therefore require shallower water for most of their growing season. High water levels would result in wetlands being more susceptible to the effects of high nutrient concentrations and may lead to the death of more sensitive species. Shallower systems also help to increase nitrification by increasing the aerobic conditions present in the cells (Carty *et al.*, 2008; Scholz, 2006).

1.2.3 Pre-treatment of wastewater

Pre-treatment is often important to achieve a better and more effective treatment of wastewaters (Cronk, 1996; Hunt and Poach, 2001). Early guidelines (USEPA, 1988) recommended the use of conventional pre-treatment units such as sedimentation basins to reduce suspended solids and biochemical oxygen demand concentrations as well as suggesting the addition of chemicals to remove phosphorus.

Constructed wetlands are often viewed as long-term solutions to wastewater treatment. One of the key aspects regarding treatment wetlands is their low cost in comparison to traditional treatment plants or other methods such as mechanical/chemical treatment. They are also seen as having a long ‘life-span’ of several decades. However, their longevity can be impeded upon by the accumulation of detritus and sediment build-up. The removal of solids in a pre-treatment stage has therefore been recommended for many years (Hunt and Poach, 2001; Cronk, 1996; USEPA, 1988). Some studies have suggested that some constructed wetland types themselves can be used as a pre-treatment step being part of a sustainable drainage system, also called best management practice in the USA (Cook *et al.*, 1996). Furthermore, the dilution of wastewaters to improve treatment is common practice in constructed wetland operation (Harrington *et al.*, 2005). This method is often just a simple addition of water to wastewater. However, heavily polluted wastewater can also be diluted by less contaminated wastewaters such as roof and yard runoff. The dilution of wastewaters is important in promoting good nutrient removal within constructed wetlands. Moreover, if the organic loading rates are excessive, this can result in decreased removal performances (Kantawanichkul *et al.*, 2003) and an increase in the risk of ammonia toxicity to some constructed wetland plants (Hunt *et al.*, 2002, 2004).

Partial nitrification of wastewater prior to its treatment in a constructed wetland has also been examined as a means of affecting nitrogen removal and ammonia volatilisation (Poach *et al.*, 2003). The most common method of additional nitrification of wastewaters is the recirculation of the wastewater itself, or the use of partially-nitrified lagoon wastewater (He *et al.*, 2006; Humenik *et al.*, 1999; Hunt and Poach, 2001; Kantawanichkul *et al.*, 2001; Poach *et al.*, 2003). Pre-treatment to achieve nitrification can reduce ammonia volatilisation due to reduced concentrations of ammonia in the wastewater (Poach *et al.*, 2003). As a result, these reduced concentrations help to minimise the potential risk of ammonia toxicity to wetland plants. The anaerobic conditions that exist in wetland soils (Scholz, 2006) limit the rate of nitrification, suggesting that denitrification in wetlands is nitrate-limited (Hunt and Poach, 2001; Hunt *et al.*, 2002, 2004).

Recirculation of wastewaters has shown to have a positive effect on total nitrogen removal across the world. Kantawanichkul (2001) showed that the recirculation of effluent in a combined vertical-flow and horizontal-flow wetland system increased the total nitrogen removal rate from 71 to 85%. Humenik (1999) reported that nitrified lagoon water added to constructed wetland microcosms led to nitrogen removal rates that were four to five times that of non-nitrified liquid.

Pilot-scale plants in China (He *et al.*, 2006) were used to test for various recirculation rates. Findings showed increased ammonia-nitrogen removal rates in comparison to non-re-circulated effluent. However, the researchers reported no great increase in phosphorus removal. This study had shown that the use of recirculation helped to form an oxide environment in the wetland.

Furthermore, the partially-nitrified state of re-circulated effluent has consistently shown its benefit to nitrogen removal in constructed wetland systems. It decreases ammonia volatilisation, which is not desirable due to it being an atmospheric pollutant (Poach *et al.*, 2004), and helps to abate ammonia toxicity with regard to the most commonly used macrophytes.

1.2.4 Sizing of wetlands

Sizing of CW is additionally highly variable and is often dependent upon the design approach taken. Vertical flow systems will predominantly have the smallest square-metre area people equivalent value (PE). These can often be as little as 1-3m^2 PE^{-1} (Cooper, 2005; Kantawanichkul, 2001), however for improved performance, they are often used in conjunction with a horizontal flow system to create a Hybrid CW which implements both vertical and horizontal design benefits. These are heavily implemented in countries were land is at a premium and saving on space is a priority (Kantawanichkul *et al.*, 2003). Horizontal flow wetlands generally have a higher land-take than vertical flow, but they can also have superior nutrient and organic material (Vymazal, 2010). The higher land-take is common throughout all horizontal flow systems, generally it is seen that 10m^2 or 20m^2 PE^{-1} is applied to various wastewater types (Carty *et al.*, 2008; Scholz *et al.*, 2007). Some sub-surface flow systems in Ireland that have worked with a smaller sizing have yielded mediocre to poor results where N and P removal have been much lower than other wetland types already in use in the country (Gill *et al*., 2011)

1.3 Macrophytes and Rural Biodiversity

1.3.1 Macrophyte types and characteristics

The range of macrophytes that are planted in constructed wetlands is wide and varied. They are intrinsic to the use of constructed wetlands and play a vital role in nutrient removal (Brix, 1994; Hunt and Poach, 2001). The USEPA (1988) guidance manual referred to cattails, reeds, rushes, bulrush and sedges, which all have different ranges of pH tolerance. For example, cattails usually tolerate pH values between 4 and 10, while other aquatic plants such as rushes and sedges have much narrower tolerance margins. The USDA (1991) guidebook on constructed wetlands lists several species that have been identified as being suitable for the use in constructed wetland systems in North America. The guidebook also states that not all wetland plants are suitable for treatment systems since they should be able to tolerate continuous flooding and exposure to high nutrient concentrations in the influent (USDA, 1991). Tanner (1996) summarized a list of properties that wetland plants should have:

- Ecological acceptability (no significant weed or disease risks, or danger to the ecological or genetic integrity of surrounding natural ecosystems);
- Tolerance of local climatic conditions, pests and diseases;
- Tolerance of pollutants and hypertrophic waterlogged conditions;
- Ready propagation, and rapid establishment, spread and growth; and
- High pollutant removal capacity.

The principal functions that macrophytes provide are numerous: stabilisation of the beds, provision of physical filtration, prevention of vertical systems becoming clogged, insulation against frost in winter and provision of a large surface area for microbial communities, which are vital to successful wastewater treatment (Brix, 1994; Scholz, 2006). In addition to supporting the treatment processes in constructed wetlands, macrophytes also provide highly under-rated aspects in traditional civil engineering design by promoting natural aesthetics and landscape integration.

Furthermore, planting of the most suitable and often native species is important in the integrated constructed wetland concept to improve the biodiversity of the vicinity around the structure (Harrington *et al.*, 2005; Scholz *et al.*, 2007). The predominantly aquatic plants provide habitats for wildlife such as mammals, birds and insects. The biodiversity of macroinvertebrates has been shown to be extremely high in certain integrated constructed wetlands in Ireland; e.g. some wetland systems have up to 60% of the countries native species of aquatic macroinvertebrates present. The adaptation of wetland plants to live in anaerobic soils is important as their root structures provide aerobic areas that help to sustain nitrifying bacteria (Brix, 1994). As well as providing oxygen for bacteria, they also provide oxygen to the anaerobic substrate and thus help to stimulate aerobic decomposition.

1.3.2 Toxicity tolerance thresholds

The toxicity tolerance thresholds and the corresponding uptake rates of pollutants by wetland plants have been researched previously (Brix 1994; Harrington, 2005; Hill *et al.*, 1997; Hubbard, Gascho and Newton, 1999). However, these studies have usually examined the more common genera used in constructed wetlands (Brix, 1994; Clarke and Baldwin, 2002; Hubbard *et al.*, 1999). For example, Clarke and Baldwin (2002) tested common species such as softrush (*Juncus effuses* L.), broadleaf arrowhead (*Sagittaria latifolia* Willd.), softstem bulrush (*Schoenoplectus tabernaemontani* C.C. Gmel.), lesser bulrush (*Typha* *angustifolia* L.), and common bulrush (*Typha latifolia* L.) at varying ammonia concentrations and water depths.

Other studies have assessed more genera growing in temperate and/or tropical climates (Tanner, 1996). However, findings concerning the effect of ammonia on plants are not fully conclusive. The preference to pre-treat the wastewater prior to it entering the wetland system or the recycling of the effluent suggests that the aquatic plants studied were most likely susceptible to ammonia toxicity, although some studies suggest that the plants are more tolerant than is commonly reported in literature, stating that there was no apparent effect on some plants due to relatively high ammonia concentrations (Hill *et al.*, 1997).

Comparisons between different plant species have been undertaken to examine their uptake rates (Hubbard *et al.*, 1999; Poach *et al.*, 2003; Tanner, 1996). Brix (1994) reported on the uptake rates of common emergent, free-floating and submerged plant species in wetlands. For example, bulrush (*Typha latifolia* L.) had an impressive nitrogen uptake rate for relatively small planted areas, but low phosphorus uptakes for considerably larger areas. The nutrients are, however, bound in the biomass, but could be removed by harvesting.

With plants being an important integral part of constructed wetlands, attention has been brought to the opportunity of using them for additional purposes. Therefore, cash crops such as soybean and rice have been assessed in terms of their use in wastewater treatment (Humenick *et al.*, 1999; Szogi, Hunt and Humenik, 2000, 2003; Szogi *et al.*, 2004).

These research studies indicated that such plants are able to grow in treatment wetlands receiving swine wastewater. The potential yield from such cash crops could make constructed wetlands using these plants more attractive, particularly in developing countries. Constructed wetlands could be used for the treatment of wastewater and also to yield a steady food supply and/or income. This would be particularly beneficial for small-scale farmers, because they would be able to produce their own feed while treating their own wastewater at the same time.

Alternative methods of using aquatic plants are not limited to cash craps. The most common macrophytes planted on floating mats in anaerobic lagoons treating swine wastewater have been assessed by Hubbard *et al.* (2004). The nutrient uptake rates were relatively high. Less commonly used plant species native to certain regions have been examined as well. For example, vetivergrass (*Vetiveria zizanioides* Nash) was used in Thailand (Kantawanichkul *et al.*, 2003). This grass was suitable for tropical hydraulic and organic loading rates.

1.4 Nutrients

1.4.1 Nutrient transformation processes

The primary objective of constructed wetlands is the removal of excessive concentrations of nutrients in wastewater. For example, the removal of nitrogen from wastewater is particularly effective by means of their design, plant assemblages and natural processes (Lee *et al.*, 2004; Poach and Hunt, 2001;Poach *et al.*, 2002; Prantner *et al.*, 2001). The removal is supported by means of filtration, sedimentation, adsorption, uptake by plants, organic matter accumulation, microbial assimilation, nitrification, denitrification and volatilisation (Brix, 1994; Poach *et al.*, 2003). However, denitrification is far more preferable than ammonia volatilisation as a means of nitrogen reduction, considering that ammonia volatilisation results in ammonia gas release, which is an atmospheric pollutant (Poach *et al.*, 2004). However, it must be kept in mind that nitrous oxide (N_2O) is released during denitrification, which is also a significant atmospheric pollutant (Lashof and Ahuja, 1990). Denitrification, however, is limited in constructed wetlands by the availability of nitrate and nitrite (Hunt and Poach, 2001; Hunt *et al.*, 2004; Hunt *et al.*, 2002). The denitrifying enzyme activity is correlated to areas where there are high amounts of nitrate and nitrite. Moreover, the denitrifying enzyme activity has been shown to increase over time with the maturity of constructed wetlands, an increased rate of nitrogen application and water depth (Hunt *et al.*, 2002). This leads to increased ammonia volatilisation. This is seen as a particular problem, especially with regard to swine wastewater and its high nutrient content. Szogi and Hunt (2001) showed that reduced water depth in a system yielded the highest N removal from the influent. They suggested that the shallower depths promoted more effective interactions in the soil-water interfaces. Their study did accept though that volatilization may have also played a role in the removal of N.

The phosphorus cycle is dissimilar to both the carbon and nitrogen cycle in that it does not involve a series of oxidation-reduction reactions. In comparison, it is predominantly a sedimentary cycle (Van der Valk, 2006).

The most common method of enhancing denitrification is to re-circulate the wastewater or to add partially-nitrified water (He *et al.*, 2006; Kantawanichkul *et al.*, 2001; Poach *et al.*, 2003). This is done by recycling the effluent back into the system or by the addition of partially nitrified storage or lagoon water. It follows that denitrification is promoted by supplying the system with greater amounts of nitrate and nitrite throughout the treatment process, thus reducing the risk of volatilisation. However, a complete removal of the volatilisation process is unrealistic.

Seasonal and temperature effects on denitrification have also been examined (Reddy *et al.*, 2001; Trias *et al.*, 2004). There is a moderate positive correlation between temperature and the removal rates in wastewaters. Trias *et al.* (2004)

reported variable findings for swine wastewater treatment with respect to total suspended solids ranging from 77% at moderate temperatures to 42% during the warmest period.

Studies with marsh-pond-marsh designs have assessed differences in the amount of ammonia volatilisation taking place in constructed wetlands (Poach *et al.*, 2004; Reddy *et al.*, 2001). While Reddy *et al.* (2001) saw similar rates of nitrogen removal to those of continuous marsh systems (often >70%) and medium phosphorus removal rates (30-45%), it was not observed how much of the nitrogen removal was associated with volatilisation.

Poach *et al.* (2002) highlighted the fact that it was not known if volatilisation of free ammonia governed nitrogen removal in constructed wetlands. Findings indicated that between 7 and 16% of the nitrogen removal was achieved by ammonia volatilisation. This gave an indication as to how much nitrogen removal was being caused by volatilisation, but also showed that it was not the principle nitrogen removal method.

Poach *et al.* (2004) used the marsh-pond-marsh design a further time to highlight the differences between the pond and marsh sections with regard to volatilisation. The pond sections had significantly higher proportions (23-36%) of volatilisation than the marsh areas (<12%). Volatilisation was the dominant nitrogen removal mechanism in the pond sections (54-79%). It follows that marsh areas should be constructed within wetland systems used for the treatment of wastewaters from confined animal operations such as swine waste and wastewater. However, relatively low removal rates of nitrogen (30%) and phosphorus (8%) have been reported for long and narrow marsh-pond-marsh systems in North Carolina, USA (Stone *et al.*, 2004).

1.4.2 Phosphorus

Phosphorus is amongst the most difficult nutrients to remove from wastewater. This has created problems for many types of treatment systems such as constructed wetlands, where it is retained but often only temporarily (Pant, Reddy and Lemon, 2001; Reddy *et al.*, 2001; Reddy *et al*, 1999; Sievers, 1997; Stone *et al.*, 2002, 2004). Similar to nitrogen, phosphorus is partly removed in constructed wetlands by plant uptake, accretions of wetland soils, microbial immobilization, retention by root bed media and precipitation in the water column (Reddy *et al.*, 2001; Scholz, 2006).

Some research studies with constructed wetlands have shown average phosphorus removal rates; 42.3-48.9% (He *et al*, 2006), 42% (Knight *et al*, 2000), 47-59% (Lee *et al*, 2006), 30-45% (Reddy *et al*, 2001), 45% (Sezerino *et al*, 2003) in comparison to some traditional treatment methods such as aeration; 65% (Suzuki *et al*, 2003), activated sludge; 47.8%. However, some other studies report very low rates; 16% – 19% (Shappell *et al.*, 2007), 8% (Stone *et al.*, 2004), 22% (Poach *et al*, 2004) Szogi *et al.*, 2004), while a relatively large proportion of investigations have shown very high removal rates; 81-94% (Hunt and Poach, 2001), 89-94% (Prantner *et al*, 2001), 98% (average of 12 systems, Harrington *et al*, 2005).

In wetland soils, phosphorus is present as soluble or insoluble, organic or inorganic complexes (Scholz and Lee, 2005). The phosphorus cycle is primarily sedimentary, though phosphine (PH_3) is released during the Phosphorus Cycle (Devai and Delaune, 1995; Gassmann et al, 1996; Hanrahan et al, 2005), and is retained in the wetland. Phosphine, which is highly flammable and indeed toxic, is readily oxidised and generally does not stay in the wetland ecosystem for extended periods of time (Hanrahan et al, 2005). Inorganic or mineralised organic phosphorus may be retained by oxyhydroxides of iron and aluminium in acidic soils and by calcium minerals in alkaline soils (Reddy and D'Angelo, 1997; Scholz and Lee, 2005). Soluble phosphorus can be removed by periphyton and subsequently by deposition of dead biomass on soil and detritus surfaces (Reddy and D'Angelo, 1997).

In aerobic conditions, insoluble phosphates are precipitated with ferric iron, calcium and aluminium (Drizo *et al.*, 2002; Scholz and Lee, 2005). This has led to experiments using various designs (vertical, horizontal, and various flow types) as well as various aggregates, such as gravel, sand, granular-activated carbon, charcoal, shale, dolomite and clay (Scholz and Xu, 2001; Pant *et al*., 2001) that are used as the primary substrate in some constructed wetlands (vertical flow systems, horizontal sub-surface flow, (Drizo *et al.*, 2002; Scholz and Xu, 2001). Several different compounds have been trialled to examine potential increases in removal rates in wetlands. Some of these materials include; Filtralite (expanded clay), BRIMAC 216 Charcoal and Electric arc furnace slag (Scholz and Xu, 2001; Drizo *et al*, 2002). The use of alternative materials to aid in the retention, adsorption and absorption of phosphorus has shown to be promising in some respects, with Electric arc furnace (EAF) slag having a phosphorus retention value of 135g P kg^{-1}. This was then shown in the same report to rise to 2.35g P kg^{-1} after being removed from the test system and allowed to rest for 4 weeks (Drizo *et al*, 2002).

The use of alternative materials may yield marginally better results in phosphorus removal, but the trade off is that of a much higher cost. Scholz and Xu (2001), compared alternative material to common filter media in a Vertical flow system as showed the cost difference to be almost 30 times that of common media in terms of materials. Such reports highlight the balance that would have to be addressed when using such materials in the construction of a treatment wetland. Their use in the treatment of swine wasteland is currently very sparse.

The partly aerobic and partly anaerobic conditions within wetlands are important for phosphorus removal. In anaerobic soils and detritus, forms of organic phosphorus are relatively resistant to enzyme hydrolysis, which would otherwise release the phosphorus back to the bioavailable pool of nutrients (Reddy and D'Angelo, 1997). In aerobic conditions, however, phosphorus is bound to organic matter and is incorporation into bacteria, algae and macrophytes (Brix, 1994; Reddy and D'Angelo, 1997; Scholz and Lee, 2002).

As phosphorus is primarily retained in the constructed wetland itself, the macrophytes or released in low concentrations as part of the wetland discharge, there has been concern amongst wetland critics with regard to the long-term ability of constructed wetlands to retain phosphorus effectively. There are several references in the literature that suggest that the larger the wetland, the better the long-term removal of phosphorus through plant uptake, accretions of wetland soils, microbial immobilization, retention by root bed media and precipitation in the water column, with physical sedimentation being the most important mechanism (Harrington *et al.*, 2005; Tanner *et al*., 1998; O' Sullivan *et al.*, 1999; Reddy *et al*., 1999; Pant *et al.*, 2001; Scholz and Lee, 2005; Carty *et al*, 2008). It follows that an appropriately sized wetland systems, such as the guidelines laid out in the SEPA (2008) document which recommend a wetland size of twice the capture area for farmyard runoff, or $20m^2$ PPE for domestic systems, have a long design life and its capacity to treat phosphorus and other nutrients can be increased (Carty *et al*, 2008; Harrington *et al*, 2005; Dunne *et al*, 2005). The harvesting of macrophytes from constructed wetlands to enhance the removal of phosphorus from various wastewaters, including swine wastewater, has been stated as both being an important method of removal as well as over-stated. Some research has shown harvesting to yield only 5% of the overall removal of phosphorus (Kim and Geary, 2001).

With many different designs (NRCS, SEPA, ICW, SYKE) yielding differing phosphorus removal rates, there is still no standardized or generally accepted design method in the literature currently available which is the most efficient at phosphorus removal. The high variability in P removal/retention rates can also be due to the characteristics of the soils used in the wetland itself. This was clearly express in Dunne *et al*, (2005), where the difference in P retention between 2 locations in Ireland was greater than double (1464mg P/kg at Dunhill, Waterford and 618mg P/kg at Johnstown Castle, Wexford).

1.5 Pathogens, Odour and Human Health

The ability of wetlands to remove pathogens and bacteria is well documented. Wetlands are similar to biofilters, which are governed by the following processes (Brix, 1994; Scholz, 2006):

- Chemical: oxidation; adsorption; exposure to toxins released by microorganisms.
- Biological: antibiosis; ingestion by nematodes, protozoans and cladocera; lytic bacteria; bacteriophage attacks; natural decay.
- Physical: filtration; sedimentation; aggregation; ultra-violet radiation.

The removal of pathogens from swine wastewater is important for human health and safety reasons. Pathogens such as *Salmonella* may enter the human body via surface waters, via drinking water or food-stuff, (Hermida *et al*, 2009; Schets *et al*, 2008; Johnson *et al*, 1995) contaminated by runoff containing traces of swine wastewater, which has been land-spread. Considering the treatment of swine wastewater with constructed wetlands, very high removal rates of enteric microbes and pathogens such as *Salmonella*, faecal coliforms and *E. coli* (Hill and Sobsey, 2001) as well as *Giardia* and *Cryptosporidium* (Karim *et al.*, 2004) have been reported.

Hill and Sobsey (2001) documented pathogen removal proportions of between 70 and 90% in surface-flow wetlands and between 80 and 99.99% in sub-surface flow wetlands with varying organic loading rates. A study into the removal of enteric pathogens showed that there were greater occurrences of *Giardia* cysts and *Cryptosporidium* oocysts in the sediments than in the water column itself. This may be indicative of the greater removal rates with regard to pathogens in sub-surface flow systems.

In Ireland, recent studies into pathogen and microbial removal rates in ICW systems have shown high removal capacities of *E.Coli* and *Enterococci* and *Salmonella* (Mc Carthy *et al*, 2011). This study examined 9 existing ICW systems, primarily treating dairy and piggery wastewaters with the recorded data concluded that such systems were capable of reducing *E. coli* and *Enterrococcus* to non-detectable levels. An average removal efficiency across 9 recorded systems showed 81.5% and 66.9% for *E. coli* and *Enterrococcus* respectively. Additionally, the tests that were performed also examined *Salmonella* in various antibiotic resistant and sensitive forms and concluded that the on-site ICW treatment systems were able to remove *Salmonella* from the wastewater being treated. Seasonal removal fluctuations were recorded in this study, showing the presence of Salmonella in the influent of some wetlands, but not in the effluent in May/June sampling periods. Such fluctuations have been previously documented in constructed wetlands treating municipal wastewaters (Zdragas, *et al*., 2002). A similar study was carried out on the

Meso-scale ICW that is described in this project and the results are covered in greater detail in section 4.5 and in McCarthy *et al*. (2011).

Odour removal in wetlands has become an important factor with regard to the construction of wetlands for agricultural wastewater treatment systems (Harrington *et al.*, 2005) and municipal wastewater (Doody *et al*, 2010) which often has a much stronger odour, or is more noticeable due to its potential proximity to domestic dwellings. The capacity for well vegetated systems to reduce malodours is important for their acceptance particularly in populated areas. Public acceptance for the use of wetlands in wastewater treatment (Smardon, 1989), irrespective of type, would allow for them to be a more appealing and viable option for use in domestic, municipal and agricultural settings.

The odour-reduction capability of livestock wastewater treatment wetlands has been primarily anecdotal (Wheeler *et al.*, 2007). However, studies have been performed to quantify odour from constructed wetlands treating swine wastewater (Huang *et al*., 2004; Wood *et al*., 2000). Findings indicated that constructed wetlands were very capable of reducing malodour from the wastewater being treated, which is seen as an additional benefit (Wheeler *et al.*, 2007; Wood & Wheeler, 2000). The suppression of mal-odours is highly relevant to the social acceptance of constructed wetlands in areas where there are residential dwellings. Despite the generally isolated location of piggeries, geographical location can affect the dispersion of odours and suppressing these is often relevant when selecting an appropriate treatment method for wastewaters.

1.6 Scope of this project

Wetlands have been part of human society for thousands of years (Maltby and Barker, 2009), but it is only in the past several decades that they have been used with specific intent and purpose for the treatment of various wastewaters. There are a range of wetland systems, regardless of their given title (e.g. constructed wetlands, treatment wetlands or integrated constructed wetlands), which have been shown to be highly efficient at treating a wide selection of wastewaters, especially those from agricultural practices. Studies from across the globe have examined aspects of their design, construction, materials, operational parameters, layout and plant species, which has resulted in improved designs and approaches on every continent where they are used. These studies have led to refinements and improvements in tested systems, allowing them to be applied to other wastewater types beyond that of their original designs, including those that deal with high-strength influents. The ICW concept has been applied to dairy, cattle, industrial, municipal and recreational (e.g. golf courses) wastewaters and surface runoff, but they have not been heavily used in the treatment of swine wastewaters. The primary scope of this project was to examine the potential optimisation of constructed wetland operations that are common aspects of various designs, from across the available literature and the refinement of these in an ICW design for the effective treatment of swine wastewaters. The additional items being examined was the practical application of a research tool (the meso-scale design) that was relatively cheap to construct, simple to operate and maintain and the performance of these systems over a full annual cycle.

Wetlands have proven themselves to be an ideal alternative option for both traditional wastewater treatment as well as a viable option for more troublesome wastewaters. This is in no small part due to their high efficiency and lower overhead costs. This in turn, is potentially an ideal alternative for the treatment of swine wastewaters, over the more conventional heavily-engineered systems. The restrictions that are being placed upon piggery farmers, such as the European Union

Nitrates Directive (S.I. No. 610, 2010) limit farmers using traditional methods of wastewater disposal such as land spreading, which are not always environmentally sustainable. The use of constructed wetlands for the treatment of swine wastewater, either as a primary treatment method or as a secondary or even tertiary treatment has been shown to give consistently high removal rates for nutrient and other pollutants as well as a reduction of distinctive malodour that is associated with swine waste. Detailed examination of a potential system that is specifically tailored to the piggery industry is greatly warranted in the context of the Irish agricultural sector as it is one of the main components of the economy. Maintaining a healthy industry will enable that industry to remain competitive. As yet, there is no widely supported alternative to land-spreading of swine wastewaters in Ireland, other than anaerobic digesters which can be prohibitively expensive. Government subsidies are not supportive enough to make them a fully viable option for many farmers (Nolan *et al*, 2012).

The use of the ICW concept and design was used for this research project as it is one of the fore-running wetland designs in Ireland, both in terms of the range of applications and treatment performance. They have been used in the treatment of various agricultural wastewaters, primarily cattle and dairy, throughout Ireland and have in the last decade, proven to be highly successful with nutrient removal rates of up to 99.9% (Scholz *et al*, 2007). Additionally, they have been used for municipal wastewater treatment, industrial waste, mine drainage as well as landfill leachate. The majority of wetlands in Ireland, prior to the introduction of ICW concept, were for filtration or polishing of wastewaters and were not used as a total treatment approach. The literature available shows the ICW approach, similar to some of the design approaches used in other countries, to be a completely inclusive design which could deal with the complex nature of swine wastewater (Babatunde, 2008; Scholz *et* *al*, 2007). Whilst there are several other designs from across the globe that could have been used in this project, the ICW concept was deemed to be the most fitting for the overall scope of the project. Simply targeting wastewater treatment is a narrow goal which does not acknowledge some of the additional benefits that many constructed/treatment wetland designs have to offer. The high level of treatment of the ICW concept, over that of other constructed wetland designs, as well as traditional wastewater treatment options (Doody *et al*, 2010) also put greater emphasis on their potential use in the role of piggery wastewater treatment. These benefits include; environmental aspects not directly linked to the quality of the wastewater (flora and fauna), social benefits and substantial economic savings. Whilst the actual design of the meso-scale takes from the ICW concept and design, it is impossible to mimic them exactly using simple equipment. A meso-scale system cannot provide social amenities, follow the RAMSAR convention or increase local biodiversity as ICW are reported as being able to provide (Harrington *et al.*, 2011; Harrington & McInnes, 2009). What the meso-scale design does provide is the creation of a potentially viable test-bed for future examination of operations and their potential incorporation into subsequent ICW implementation.

Originally, the remit of this research project was to examine the treatment of raw swine wastewater, but this was subsequently changed to the treatment of anaerobically digested swine wastewaters. This was changed as the PhD project was combined with the the Irish Department of Agriculture Food and Fisheries' Research Stimulus Fund Programme (RSFP) under the National Development Plan 2007-2013 as well as being funded by the scheme. The Stimulus Research group was operating a project entitled "Energy generation option for pig manure and sustainable disposal of residue", which encompassed experimental anaerobic digester design, composting of solids, woodchip filters, pyrolysis, constructed wetlands and microbial analysis of all techniques. Treatments were divided into solid and liquid groups, with the woodchip and constructed wetlands were those that examined the liquid fraction of the anaerobically digested material, though the material was not that which was produced from the experimental digester, it was sourced from a piggery in the south-west of Ireland. The RSFP provided research funding for the duration of this project.

The operations that were chosen for this research, which are described in depth in Section 2, are common to the vast majority of constructed and treatment wetlands around the world, regardless of their primary wetland type (sub-surface flow, vertical, surface-flow). Whilst the primary design is based heavily upon the ICW concept, pioneered in Ireland, the examined operations and indeed some of the design features and mechanics are common throughout treatment and constructed wetland designs worldwide. Each have variations on flow mechanics and loading (both hydraulic and nutrient) rates, which are tailored for the influent being treated as well as the climatic and geographical conditions under which the wetland is found. These shared attributes between the ICW and other CW designs show their importance with design and the effects that variation and adjusting of simple integral features can have on the performance of CW. Focusing on these shared parameters and comparing the relatively new constructed wetlands being used in Ireland allow for closer cross-examination with those that have been in use for lengthier periods of time elsewhere in the world. As per the available literature, the first occurrences of constructed wetlands in Ireland began in the early 1990's (Babatunde, 2008), whereas in other regions of the world, wetlands had been catalogued for at least a decade before that, with the highest number of catalogued treatment wetlands in the United States (USEPA, 2000). Having these shared features/operations was deemed to be important as it would allow for potentially easier application of any findings based upon the approach used in this research project.

The examination of a potentially practical, affordable, sustainable and environmentally beneficial approach to piggery wastewater management is of great importance in the current environmental and financial situation that the world currently finds itself. This research project aims to examine how the ICW concept and approach may be applied to the piggery industry in Ireland and if the results obtained yield additional information for other industries and fields of research. Additionally, it seeks to examine the applicability of the meso-scale design as a viable and practical test-bed for the examination of wastewater treatment through an adaptable constructed wetland design.

The thesis will test specifically the following;

- The performance of specific operational parameters in an experimental ICW in order to examine any potential superiority of a single parameter when treating a known influent.
- The potential performance of ICW systems to treat the liquid fraction of anaerobically digested swine wastewaters.
- The practicality and feasibility of an experimental meso-scale surface-flow wetland in its capacity to mimic full-scale wetlands based on the ICW concept and design as well as being a practical research tool.

2. Materials and method

This section covers the location, materials, construction methods and operational parameters of the 16 meso-scale systems that were developed in Moorepark Teagasc Research Centre in Ireland. Throughout the operation of this project, all construction (where operated machinery was not required), installation, planting, maintenance of both the site and equipment, sampling and laboratory analysis (except that of BOD_5 analysis) was performed by the student. The range of analytical work that was performed was not limited by the lack of additional help, but rather that of budgetary restrictions. Initial funding that was approved was reduced by 66%, this placed restrictions on the range of parameters that could be analysed. The range of parameters is explained in detail in Section 2.8, these were chosen as they are the similar range of physico-chemical parameters that are analysed on full-scale ICW systems in SE Ireland in Co. Waterford. This allowed for relative comparisons between the meso-scale systems and full-scale ICW systems that have been in operation for several years.

2.1 Site location and description

The meso-scale ICW sits on a gently sloping, NE-facing field (Fig 1.) in Moorepark Teagasc Research Centre, Fermoy, Co. Cork in the south of Ireland. The site location was decided due to a suitable slope and access to necessary supplies of electricity and water (Fig 1). The mean annual temperature is about 11.5 ^{o}C and the mean annual rainfall is roughly 1150 mm. The approximate mean seasonal temperatures for the region were as follows: winter, 8.0 ^{o}C; spring, 10.5 ^{o}C; summer, 15.0 ^{o}C; and autumn, 12.5 ^{o}C (Met Éireann, 2010).

2.2 Basis for the systems and their design

The basic core of the design was the implementation of the Integrated Constructed Wetland (ICW) concept. The primary sources of literature were Harrington and Ryder (2002) and Scholz *et al*. (2007). From these 2 key papers, it was highlighted that the “cells” of an ICW should have a Length: Width ratio of less than 2:1 with a shallow operational water depth of less than 30cm. The design also highlights that influent into the systems and the influent flow should have the longest path to travel that the cells allow. This allows for greater interaction with the micro/macro biota within the cells and for a longer retention time.

Fig 1. Satellite view of research site in Moorepark Research Centre, Fermoy, Co. Cork with Meso-scale wetland area outlined in white.

Fig 2. Initial trenches dug on site.

The Meso-scale ICW was designed to be simple and almost completely autonomous. This was to rule out as much human error as was possible during operation. The layout of the systems was designed to be incorporated into the site that was made available in Moorepark, Fermoy. The site had a gentle slope that allowed for the systems to be embedded into the site below ground level. The individual systems were constructed using readily available polyethylene containers, braided garden/industrial hosing, 60mm wavin ducting and in-line submersible DC pumps.

As described in Section 1.6, these meso-scale systems are designed to replicate intrinsic design parameters shared with many CW designs, but moreso those of the ICW concept, in terms of the number of cells, nutrient and hydraulic loading rates, aspect ratio, free surface water design, macrophyte selection, water depth and low overhead/maintenance costs.

The meso-scale system comprises of 16 small-scale wetland systems, all built using 60-litre polyethylene containers measuring;

External: L 600 x W 400 x H319mm

Internal: L 555 x W 355 x H300mm

Each wetland system comprised of 5 cells. The first cell received influent from a storage container, which flowed through the wetland system sequentially. The initial four cells are treatment cells with the fifth cell being a collection cell for any outflow. The containers used, provided an adequate length: width ratio and supported the shallow water depth requirement of the ICW design.

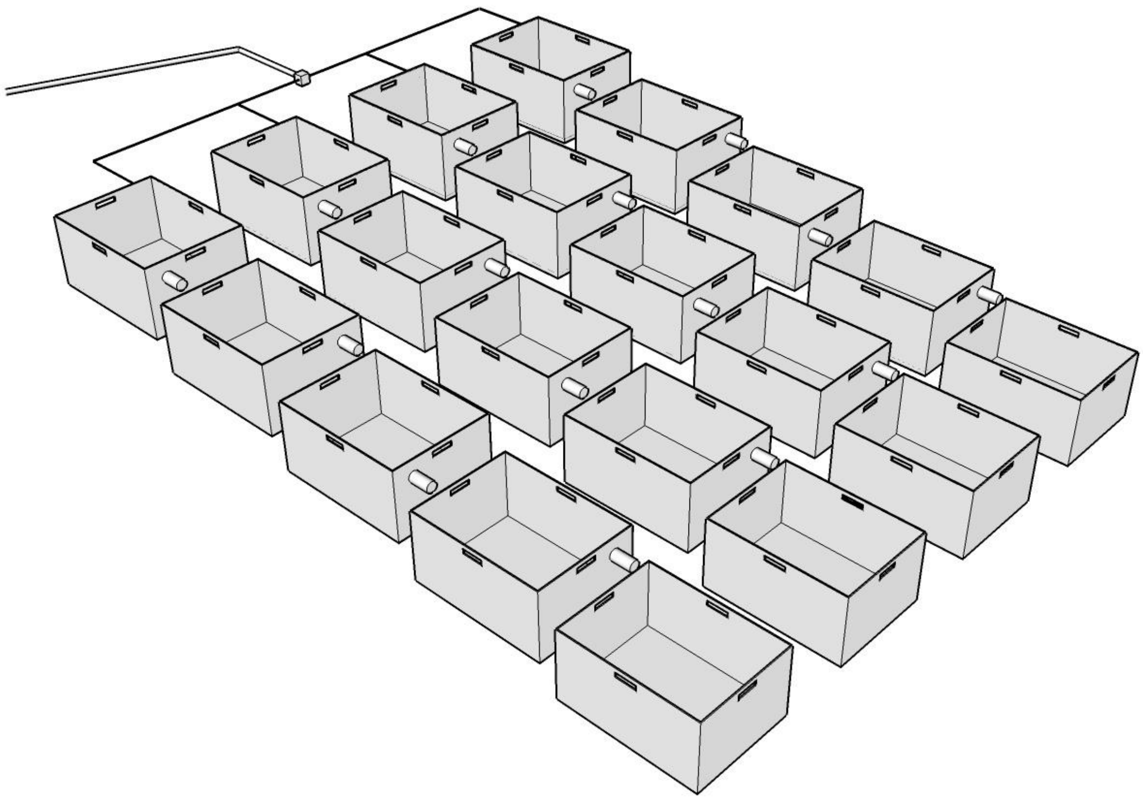

Fig 3. Schematic model of one operational set, with 4 replicate systems.

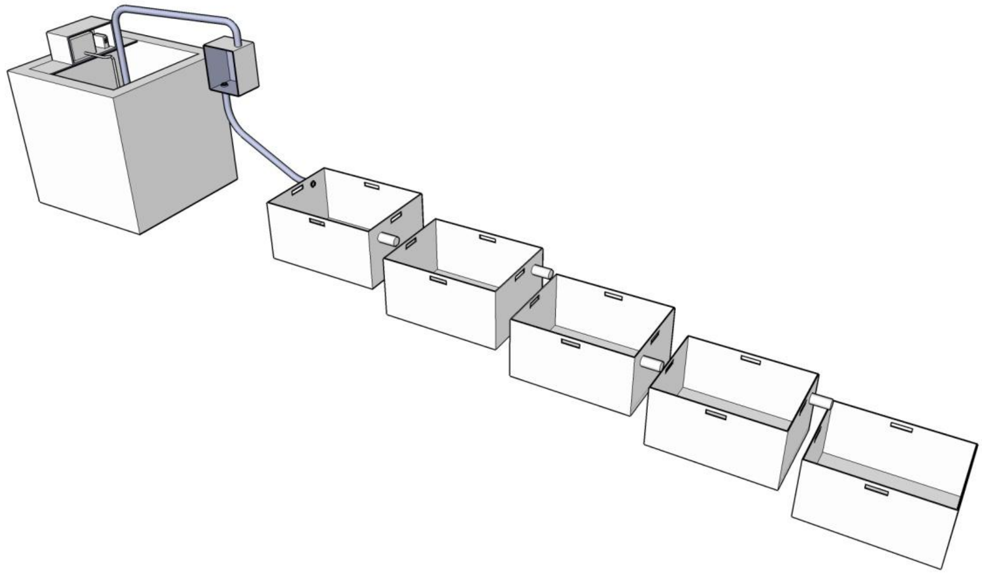

Fig 4. Schematic model of a single system connected to the IBC container with in-line pump and siphon trap installed.

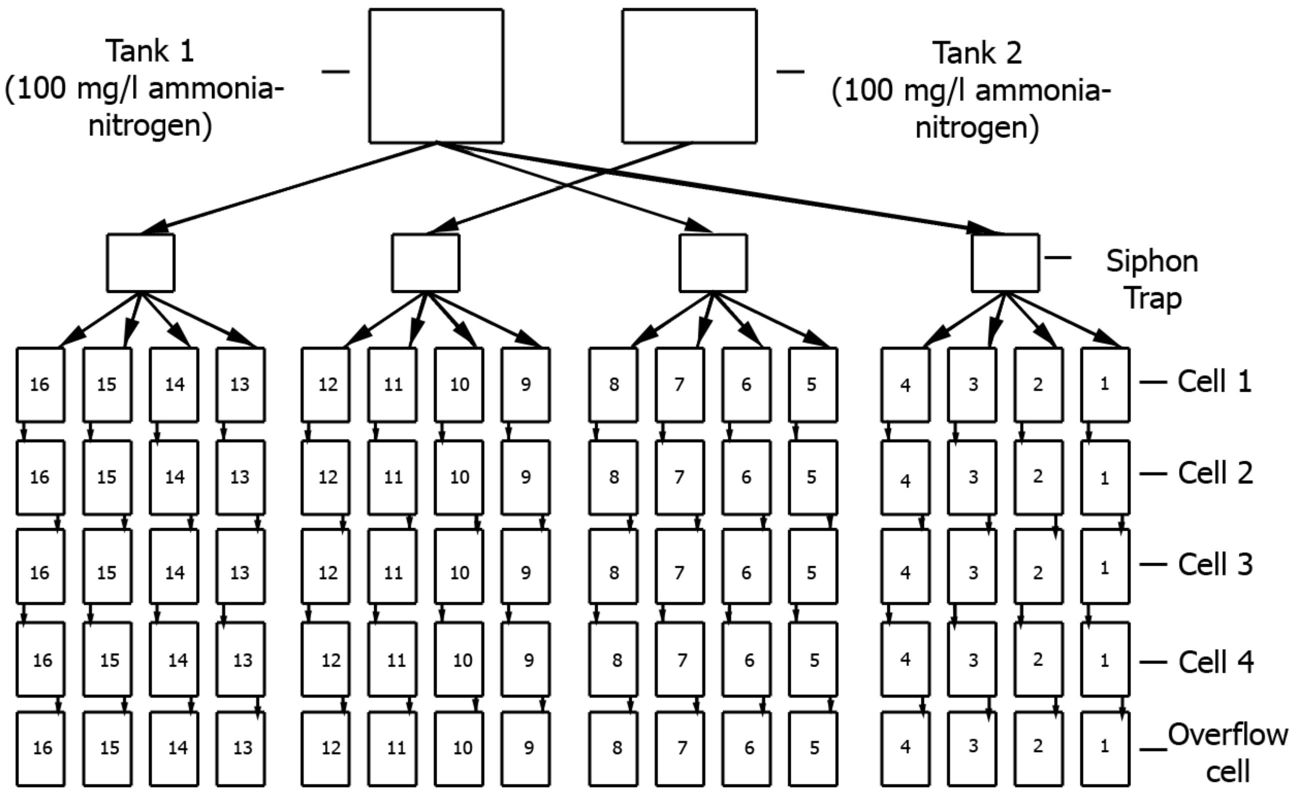

Fig 5. Diagrammatical view of the full layout of the meso-scale systems with flows indicated by arrows.

2.3 Construction

2.3.1 Construction methods

The meso-scale systems were initially dug using a mini-digger to excavate tiered trenches to allow the cells to be kept below ground level (Fig. 4). This was important to reduce the amount of direct sunlight on the systems as well as to attempt to maintain "embankment" temperatures the same as the surrounding soil. This is again a part of mimicking the ICW design in regard to their earthen embankments. Additionally, this also helps to maintain temperature homogeneity across each system and its replicates. Within a relatively small area, the ground temperature is similar and more stable across all of the replicate systems.

Fig 6. Polyethylene containers laid out in tiered trenches.

Overflow pipes were connected at adjacent corners of each box using a hole-saw and heat gun. The holes were drilled at a smaller diameter than that of the wavin ducting and the edges were heated to make the sides malleable. The ducting was cut into 20cm lengths and forced through the hole whilst the polyethylene was soft. Each overflow pipe was fitted 2cm below the level of the handles that are on all 4 sides of the polyethylene containers. Once the polyethylene had cooled and hardened, a soft outdoor silicone sealant was applied around the outlet pipe on the interior surface of the container (Figs 5 & 6.). This sealant was applied to the containers in a clean room and all surfaces had been cleaned and prepped accordingly so as to minimise the chances of dirt or particulate matter having a negative effect on the seal during curing or usage. After being fitted, the containers were stored indoors for 24 hours while the sealant cured. The containers were then placed into their trenches (Fig 7.) and each cell was checked to be level. The overflow pipes for each cell were laid out in an alternating pattern so as to maximise the distance between the inlet and outlet of each system. This helps to increase the hydraulic retention time within each cell as well as reducing the risk of preferential flows through each cell.

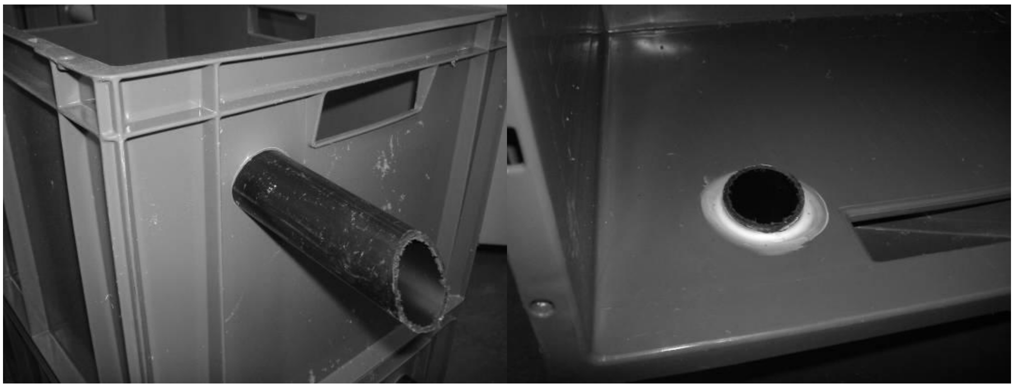

Fig 7 & 8. Overflow pipes made from waving ducting, which was sealed after fitting.

The tiers within each trench were checked during construction for level sitting and areas that were uneven were corrected using in-situ soil and rough 6cm aggregate to obtain a reasonable level. The containers were filled with water after the addition of aggregate as a substrate and planted. The following week, levels were checked again and corrections made to each cell if needed.

2.3.2 Homogeneity of systems and the key variables

In order to decrease any variability between and indeed within each system, it was decided that a washed, rough (~6cm) aggregate would be used as the substrate within each container. Soil variability has been shown to have effect on the performance of nutrient removal (Dunne *et al*, 2002), especially with regard to phosphorus species retention/capture. While increasing performance would be beneficial, maintaining or even achieving homogeneity across 16 systems would be nigh-on impossible if using a soil-based liner.

Rough aggregate was sourced and brought on-site in a single delivery, washed thoroughly by hand and added to each system to a depth of 10cm in each cell. This limestone aggregate was sourced from a Roadstone supplier and was a 50mm crushed rock aggregate. The aggregate was measured in each container to obtain similar aggregate depths (c. 10cm) and water depths and then levelled off.

Fig 9. Aggregate added to single cell.

2.4 Macrophytes

Glyceria maxima (Reed sweetgrass) was chosen as the primary macrophyte to be used in the Meso-scale systems. It was chosen due to its rapid growth and expansion, high uptake rates of nutrients, tolerance of nutrient levels and sustainability. Primarily, *Carex riparia* is used as the dominant macrophyte in constructed wetland systems due to its being evergreen. However, it has relatively low tolerance for high levels of nutrients (primarily ammonia) or low tolerance over extended periods. It does not colonise available area as quickly as Glyceria and its colonisation can often be impeded by nutrient levels.

Glyceria maxima is not commonly used as the primary macrophyte in constructed wetland studies, though it has been used in a great number of studies. This may be due to various factors including tolerances, climatic conditions as well as it being deciduous. It is a species of rhizomatous perennial grasses in the mannagrass genus native to Europe and Western Siberia and growing in wet areas such as riverbanks and ponds.

Mature (1 full season of growth), bare-root Glyceria was planted in numbers of 50 into each cell (255/m^2). This was to provide overall coverage of each cell with healthy plants to promote rapid colonisation. The mature plants were sourced from an existing wetland that was no longer in operation in Kilmeaden, Co. Waterford. These plants had been used in a large, industrial wetland that had been treating effluent from the Kilmeaden Cheese factory, though the site had not been in operation for several years when the plants were harvested. There had been a 5 year gap between harvesting of the Glyceria and when they had last been in contact with any wastewater. As such, they had not been introduced to nutrient-rich influent prior to the startup of this project. The planting occurred in August of 2008 with all cells being planted with 24 hours. Each cell was then filled with tap water to a depth of

15cm in order to support and maintain the plants. The plants were not fed any soiled or nutrient-rich water so as not to impede their growth and acclimatisation to the low-nutrient levels. High levels of ammonia-N being fed into the system too early into the establishment of the plants can be detrimental to their establishment and stunt growth or indeed kill the plants completely. This has been seen in many constructed wetland systems across the globe where an “Ammonia burn” kills the plant life in a cell in a path and can create preferential flow through a cell, or a full system and greatly reduces the hydraulic retention time.

Fig 10. Mature *Glyceria maxima* plants added to cells before adding water.

While the plants and systems were establishing, a plastic garden shed on a concrete foundation was erected to house the 1m^3 storage tanks which would contain the influent for the systems. The shed itself measured 1.4m x 1.4m x 2.2m. Into this, two 1m^3 were placed as well as a 12v DC convertor for the submersible pumps.

Fig 11. Shed installed and operational with storage tanks and pumps

2.5 Startup

Self-priming submersible pumps were used for the addition of influent to the meso-scale systems. These were marine pumps commonly found on small vessels for the use of pumping storage water about the ship. These were tested for pumping rates and found to produce very similar rates of flow with tolerances ±20ml/min. These pumps were held in the storage tanks, held at the centre by means of the electrical cables that powered them and 13mm (internal diameter) braided tubing.

13mm braided tubing was used leading from the pumps themselves to Koolance® 4-way tube splitters (Fig. 11). These splitters diverged the 13mm tubing to 4 x 6mm (internal diameter) tubing that led to each system. This resulted in 4 pumps supplying the 4 treatments being examined. 16 systems fed by 4 pumps, each pump supplying the material for its own specific treatment operation. This helped to minimise any problems with layout, faults, breakdowns and general maintenance with each system due to their being independent of one another.

Fig 12. Tanks installed in shed

Each pump produced 11.75 l/min. This was split into 4 pathways, delivering 2.94 l/min. The initial application of influent was worked out using a joining of a baseline influent level of 100m^3/ha/d, which is often used in dairy ICW systems in Ireland. Taking this application rate of 100m^3/ha/d and scaling it linearly down to the surface area (0.197m^2) of the containers being used in each system, an application rate of 7.88 litres per day is achieved. Whilst a somewhat simplistic approach, it was deemed the most reasonable approach to take when the meso-scale was being designed and developed.

$$100\text{m}^3/\text{ha/d} = 100{,}000\text{L} / 100\text{m}^2$$

$$10\ \text{L/m}^2$$

$$\text{Box area} = 0.197\text{m}^2$$

$$\text{System area (cell x4)} = 0.788\text{m}^2$$

$$= 7.88\ \text{L/day}$$

Fig 13. Koolance 4-way tube splitters used to split flow from 13mm to 6mm

These pumps were connected to electrical timers to regulate the pumping timetable to as to rule out human error in the application of material to the systems. The timers that were originally used were insufficient as they would not allow single-minute operation of the pumps and were replaced with industrial timers. These updated timers also had security coding to prevent any possible tampering or accidental changes to the pumping times.

Fig 14. Updated timers fitted to pumping mechanism with master switch

Upon startup, both the timers and pumps performed as expected. However, due to the placement of the piping and their orientation below the maximum level of the stored liquid, a siphon effect was produced and this resulted in the addition of excess material to the systems even as the pumps were switched off. This was fixed by the use of 5L containers to act as siphon traps (Figure 15). These were attached to the inner wall of the housing shed and set upon assembled shelving. The tubing from the tube-splitters fed into these containers through loose-fitting holes in the top of the containers. The material that entered the container flowed out of 6mm tubing that was connected through the cap of the container which provided a form of seal due to the expansion of the rubber tubing. This was further sealed using clear silicone that was applied to the outside of the cap thread and the outside of the connection of the hose to the cap.

Fig 15. Siphon traps installed in shed

A fifth overflow cell was added to each system in order to record outflow volumes from each system. These were the same size as the other cells in each system and were taken from excess stock from what was initially ordered during construction. Fitted lids were ordered and a 6mm (internal diameter) braided hose was fitted to each container to accept any outflow from the systems. This outflow hose was fitted below the levels of the handles on the boxes so that the outflow material would preferentially flow through it.

Sediment from the influent material was present, but due to the system design focused on the liquid fraction from an anaerobic digester, the particulate matter was generally very fine in nature and rarely caused any blockages. Blockages did occur in periods of exceptionally low temperatures when accumulated sediment became dislodged and blocked the outflow pipe from the siphon traps. This is discussed further in sections 2.10 and 4.9.

2.6 Operational methods

Key operational parameters were identified through the available literature at the beginning of this project and it was decided that the observed key parameters were that of;

- Nutrient concentration
- Influent volume
- Recycling of effluent

Each of these parameters was controllable directly through timings on the pumps themselves. This helped to maintain the design criteria regarding ease of use and autonomy. With set influent volumes from each pump over a 1-minute interval, loadings could easily be adjusted and changed when needed. Each submersible pump is capable of pumping 11.75 L/min at 1 metre, with the equal distribution through the Koolance splitters due to an internal conical divider. The pumps are capable of delivering 2.94 L/min to each replicate. The initial loading set out from the container area of 7.88 L/d did however result in the timings and loading not matching exactly, with the timers being limited to 1-minute timings. Coupled with the delivery of 2.94 L/min, 3 minutes of pumping was decided upon as the base loading rates.

This gives a slightly increased loading when scaled up to full-scale operation of 112m^3/ha/d. This increased influent loading was not considered to be an issue and that it would not impact negatively on the systems. The pumping of material into each system was done over 3 intervals of 1 minute pumpings each day.

$$\text{Pump} = 11.75\text{L/min}$$

$$11.75\text{L} \div 4 = 2.94\text{L/min}$$

$$7.88 \div 2.94 = 2.6 \text{ minutes} \Rightarrow 156 \text{ seconds} \Rightarrow 2 \text{ minutes } 36 \text{ seconds} \Rightarrow 3 \text{ minutes}$$

$$\text{Pumping of 3 minutes} = 8.82\text{L/day} = 1.119 \times 100\text{m}^3\text{/ha/d}$$

$$\Rightarrow 112\text{m}^3\text{/ha/d}$$

This initial loading rate was initially performed from September 2008 to December 2008 with tap water from the two 1m^3 storage tanks. From December 2008 until March 2009, Tank 1 and Tank 2 were given quantities of separated liquid from an anaerobic digestor. This was to raise the ammonia concentrations to roughly 100 and 200 mg/l respectively. The liquid being added was tested the previous week for the ammonia value and once results were confirmed, it was added at set ratios to the volume of tap water in the storage tanks and mixed thoroughly by hand.

These concentrations and volumes were maintained from December 2008 until March 2009. Initial samples had shown high removal rates and treatment efficiency, but at the end of February removal rates had dropped to less than 10%. It was decided that the influent loading volumes would be dropped to 66%. This was achieved simply by decreasing the number of 1-minute pumpings to the systems from 3 to 2 daily. However, this did not yield substantial results and in April 2009, the volumes were decreased to 33% of the original. These loading volumes were maintained and resulted in the volumes entering the system being as follows;

$$\text{Pump} = 11.75\text{L/min}$$

$$11.75\text{L} \div 4 = 2.94\text{L/min}$$

$$\text{Pumping of 1 minute} = 2.94\text{L/day}$$

$$2.94 \text{ X } 1.27 = 3.7338 \text{ L /m}^2\text{/day}$$

$$= \mathbf{37.3m^3/ha/d}$$

In the case of the High Flow Rate, the pumps ran twice a day, resulting in an influent volume equivalent of $74.6m^3$/ha/d. These influent volume rates were maintained for the following 12 months for which sampling was run, June 6^{th} 2009 to June 10^{th} 2010.

2.7 Sampling methods

The Sampling of each and all systems was done weekly, each Wednesday from 10^{th} of December 2009 until the 6^{th} of June 2010. Each sample was taken using sterile gloves and with 500ml sealable containers. Normally, samples would not need to be of this size for nutrient analysis, but the requirement for BOD_5 analysis demanded at least 250ml per sample and often analysis would need to be performed twice, so additional material was needed.

Each sample was taken from immediately before the outflow of cells 1 and 4. This gave the most accurate, and reliable representation of the outflow material from each cell. In instances where there was no outflow from cell 4, a sample was taken from cell 4 as close to the outlet as possible and every effort was made to keep the level of sediment disturbance to a minimum. Sediment entering the sample would have the potential to skew the results of any analysis performed.

Sampling was performed alongside all maintenance that needed to be performed on the site, but always beforehand. Samples were also taken from the

Storage tanks for nutrient analysis, primarily Ammonia. This was done in order to maintain the ammonia concentration levels.

All 16 systems were sampling in the same order each week and samples were labelled with a Researcher Code, designated by Teagasc Research centre, Johnstown Castle in Co. Wexford. The code given was OJD (1 – 2,400). This code was required for accurate recording and filing of samples with regard to BOD5 analysis. The implementation of this code also allowed for more streamlined cross-referencing of samples if any errors were reported. The storage tank samples and any additional samples were not labelled with OJD codes and were filed separately.

When each batch of 34 samples was taken (32 system samples and 2 storage tank samples), a Orion 5-Star Multimeter hand probe was used on the samples to record pH, Conductivity, Temperature and Dissolved Oxygen. The probe was calibrated before each sampling set. The calibration was done using field calibration kits immediately before samples were collected. Each 500ml sample that was taken was kept in a carrying container during delivery to the laboratory where analysis was performed. Immediately after the samples were taken and placed in the container, the Orion probe was used to analysis each sample in sequence. After each cell was analysed for the multimeter parameters, it was rinsed in de-ionised water before being placed in the next sample. The use of the multimeter sampling took on average 30 minutes to complete and temperature increases were kept to a minimum.

2.8 Sample analysis

Physico-chemical sample analysis was performed at the Water Research Laboratory, Waterford County Council, Adamstown, Co. Waterford and Teagasc, Johnstown Castle Research Centre, Co. Wexford.

Samples in the Adamstown Laboratory were analysed for ammonia nitrogen (NH_3+), molybdate reactive phosphorus (MRP), nitrate, nitrite, total oxidised nitrogen (TON), and chloride. Biochemical oxygen demand (BOD5) was analysed in Johnstown Castle water research laboratory.

Samples were analysed on the same day as fieldwork, or within 24 hours of them being taken. If they could not be analysed the same day, they were kept in a fridge overnight and brought to the laboratories immediately the morning after. From each 500ml container, a maximum of 2ml was taken for the vials which were used in the Kone analyser in Waterford. The remaining material would then be transported to Johnstown Castle for BOD analysis every 2nd week. This was to help minimise the amount of materials needed for fieldwork.

All analysis was done in accordance with the 'Standard methods for water and wastewater' published by the American Public Health Association (APHA), American Water Works Association (AWWA) and Water Environment Federation (WEF).

Parameters that were analysed in Adamstown we taken from 2ml samples of the effluent from cells 1 and 4. Additionally, material from Storage tank 1 and 2 as well as the raw separated digestate liquid were analysed under the same nutrient parameters. The samples from storage tank 1 & 2, raw digestate and cell 1 were regularly diluted with dionised water in a ratio of 1:8. This dilution was done by hand and recorded on the Kone analyser so as to prevent any “drift” due to high ammonia concentration on the apparatus.

The analysis performed by the Kone is primarily SPECTROMETRY and incorporates the following methods;

1. Chloride: Ferricyanide method
2. Nitrite: Colorimetric method
3. Total oxidised nitrogen (TON) - Hydrazine reduction method
4. Nitrate: by subtraction of nitrite from TON
5. MRP- ascorbic acid method

Biochemical oxygen demand (BOD_5) analysis was performed in Johnstown Castle Teagasc Research Centre on a fortnightly basis. Analysis was performed only on the cell 1 and cell 4 samples due to budgetary restrictions. BOD was analysed according to standard methods, using a WTW dissolved oxygen (DO) probe. Residual dissolved oxygen (DO) was subtracted from initial DO after five days incubation at 20°C, following suitable dilution with nutrient buffer solution (Baird, 2005).

All data that was collected from the meso-scale system was entered into Microsoft Excel for organisation and basic analysis and graphical representation. Each system and the replicates included therein, were organised and categorised by date and physico-chemical parameter (Appendix 5). Statistical analysis was performed in conjunction with Favel Naulty (University College Dublin) and done SPSS Statistics software (Version 17). The data was organised, discussed and laid out in a manner that made using SPSS the most practical software for the statistical analysis of the complete dataset. Results of this analysis is located in Section 3 (Results and Discussion).

2.9 Additional sampling topics

In addition to the regular sampling of cells 1 and 4, storage tanks 1 and 2 and the raw material being utilized, there were changes made to the design and sampling procedure. A 5th cell was added to each system as an overflow collection unit. This cell was identical to those used in the project, but there was no aggregate used or macrophytes.

The 5th cell was added to the systems on 4th/April/2009. They were placed below ground level as with the other cells. This was simply to ensure gravity flow of any outflow material into the collection cell. The cell was connected to cell 4 with a 6mm/13mm (internal/external diameter) braided hose that was set into the side of the cell through a drilled hole. This outlet was secured simply by using a smaller diameter hole to ensure that no material would exit the system through means of leaking.

Initially, these hoses fed directly into 5-litre containers that were placed inside the cells for collection of effluent during periods of low or no flow. These containers were removed later as rainfall increased later in the year. The containers were also given a fitted lid to prevent rainfall entering the collection cells so as to improve the accuracy of the measurements of recorded outflow.

The sampling of the outflow material was done on-site the same day nutrient samples were taken. Measuring of the collected flow was done initially by using a graduated bucket but was subsequently changed to a simple depth measurement in centimetres. With a fixed area in each container, a volume was easily calculated based upon the depth of the effluent inside the collection cell. Each cell has an internal surface area of 0.197m^2. When the depth was measured in the cell, 15cm for example, it would yield a volumetric result.

$$0.197 \times 0.15 = 0.02955m^3$$

$$0.02955m^3 = 29.55 \text{ litres}$$

The recording of outflow volumes was started in late April once the fitted lids had been obtained for the cells and continued through the summer and into the autumn of 2009. The sampling was not done as regularly and consistently as the nutrient sampling. Additionally, there was a period during the summer months where zero outflow was recorded in several of the systems. In some replicates, there was zero outflow for several months, predominantly in the Recycling systems where their longer retention time would have an effect on outflow due to increased evapotranspiration. The implementation of outflow volume measurements was not initially incorporated into the design, similar to the overflow containers and a sampling protocol was not implemented correctly. Some readings were taken when fieldwork was being conducted, but it was irregular throughout the sampling period. During periods of heavy rainfall, the outflow

Between April 2009 and May 2010, a microbial study was performed on 2 replicates from each treatment method by Gemma McCarthy of Waterford Institute of Technology. The study examined samples for the presence of *Escherichia coli*, *Coliforms*, *Enterrococcus*, yeasts and moulds and spore-forming bacteria. The absence/presence of Salmonella was also examined in the raw, influent and effluent of the meso-scale systems (McCarthy, *et al.*, 2011). Initial sampling of the meso-scale systems was performed on a single replicate from each treatment for the first 3 months. Samples were taken from each cell and both of the storage units as well as the raw material prior to dilution. Sampling was increased to 2 replicates from each treatment after this initial 3 month period. Over the total 13 months of sampling, one sampling period was missed in January of 2010 due to adverse weather conditions where each of the cells was frozen over. The results of this study are summarized in Section 3 and are covered in depth in McCarthy *et al*. (2011). The findings of this paper are best viewed in comparison with the findings of a similar study operated at the same time on full-scale ICW systems that were in use treating dairy and swine wastewaters (McCarthy *et al*., 2011). This study examined 9 other full-scale ICW systems that had been in operation in County Waterford, 3 of which were treating piggery wastewaters and 6 which were treating dairy wastewaters. This larger study focused on indicator micro-organisms *E. Coli, Enterrococcus* and *Salmonella*. This study sampled these large-scale systems on either March and May 2010, or May and June 2010. The results of these two studies are briefly discussed in section 4.5 and in-depth in their respective journals.

2.10 Problems found and solutions

Throughout the research, design, construction, fieldwork and operation, certain problems and errors were noted. Some were corrected and dealt with as effectively as was possible, others were of a manner that they had to be adapted to and overcome. Whilst none of these had a dramatic impact on the meso-scale systems, they were unforeseen and not expected in the earlier stages of the design.

Initial problems with the design itself were that of the pumping apparatus for each of the treatment system replicates. Upon first operational testing of the pumps, transformer and splitters, a siphoning effect was noted from all 4 pumps. This was due to the pumps removing any air that was in the pipes feeding into each system by simply pushing the air out. With the receiving cells being roughly 60cm lower than the intake of the pumps, a siphon effect occurred and once the pumps switched off after their programmed operation, the influent material continued to flow non-stop into each system. This occurred only once upon initial startup of the systems.

The siphon effect was overcome using a series of 3-litre inverted containers into which each delivery hose ran into from the splitters. This allowed the influent to enter the container during pumping and then leave the container and flow to the receiving cell by a gravity feed. An inlet hose from the tube splitters entered the container at the top rear and an outlet hose was attached into the cap of the

Fig 14. Image showing the installation of siphon traps with storage tank feed lines.

container. Each cap was sealed with silicone and leak-tested. With air being present in the containers, there were never any siphoning problems since. Figure 14 below shows the feed lines (13mm) from storage tanks to the Koolance tube splitters which then diverge into the 4 lines (6mm) which feed into the siphon traps. The Koolance splitters were positioned such that the 13mm feed from the storage unit would be near-vertical in order to ensure even distribution through all 4 siphon trap lines. This was done by fixing the storage tank lines to the support struts in the roof of the storage shed.

Algal growth in the cells where macrophyte density was lower, due to lower nutrient levels which impeded their growth, the outflow hose from cell 4 to the collection cell would occasionally become clogged and prevent outflow from going into the collection cell. The result from this was that the overflow from cell 4 would flow out via the handle openings in the side of the cell and the material lost.

This was observed and dealt with when noticed in each cell when weekly sampling was being performed. Excess algal growth around the effluent pipe was removed, but only that immediately around the hose itself. This was to prevent disturbing the material in the fourth cell unnecessarily. While it did not always prevent blockages happening, it did help to maintain reliable flow to the collection cells. These blockages were sporadic and irregular, occurring primarily in spring months. The amount of liquid that had spilled over the handles of each of the containers is unknown, but from visual inspection each week, it is thought that it was minimal. This is based on the small amount of algae present blocking the pipe, the low flow and the general lack of heavily saturated soils immediately surrounding the cell where liquid would have fallen. It is estimated that the volume from such incidents would have been <500ml over a 7 day period.

Extreme weather was an issue in the winter of both 2009 and 2010. The winters over the study period had record low temperatures throughout Ireland for the past 40 years. In Fermoy, Co. Cork where the site is situated, the temperatures reached levels where the cells in each system, even those with dense amounts of macrophyte growth and detritus were frozen. On one occasion the ice was recorded at 60mm thickness. Ground temperature averages for January 2010 were at 2.0, 2.5

and 3.1 at depths of 100mm, 200mm and 300mm respectively (Met Éireann, 2012). With the sides of the containers set below ground level and the container sides exposed to the elements, the temperature drops were sufficient to cause the containers to freeze heavily.

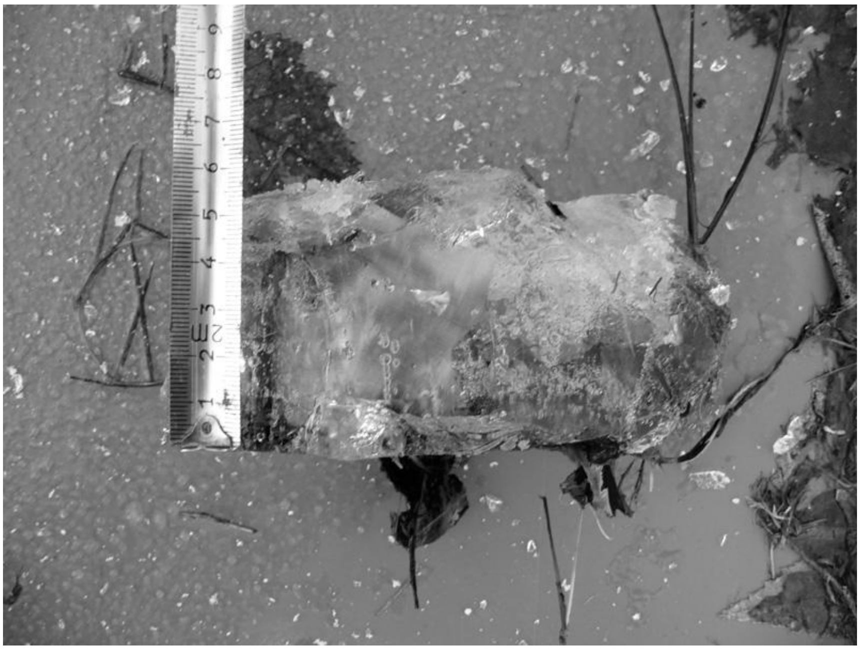

Fig 15. 60mm ice taken from Cell 4 of Normal 1 on Jan 6, 2010

When the systems were fully frozen, samples were not taken from any of the treatment systems. The ice thickness was recorded in places, but as carefully as possible to minimize disturbance to the cells.

When the weather was cold enough to result in a constant frozen state in the cells, the electricity to the pumps was switched off. The internal lighting in the storage shed was left on, but otherwise all electrical items were powered down. This was to try to prevent any damage being done to the pumps if the stored liquid would freeze to a point where the pumps would not receive liquid. Additionally, there was no need in supplying liquid into the cells if they were frozen as samples would not be taken.

The winter of 2009/2010 was severe enough that the material housed in the storage tanks also began to freeze. While they did not freeze solid, the liquid stored became frozen enough that 3 of the 4 submersible pumps were cracked due to the pressure and/or burnt out motors. The submersible pumps were self-priming pumps and with the liquid being in a state that it could not be pumped through the pumps, they were unable to be primed and most likely caused the motors to burn out. These pumps were replaced once the entire system was no longer frozen and fully operational again. Replacement pumps had been purchased during the initial build process of the project and kept for redundancy and replacement purposes. All of the electrical wiring was inspected and checked for potential faults. New wiring was used on the pumps as well to ensure good connection to the timers and power converter. Inspections were done on the pumps themselves and attempted repairs, but these were unsuccessful due to lack of expertise and no replacement parts. This was attempted in order to see if the pumps could be restored to operational quality and to keep the theoretical costs down.

2.11 Overall status.

Generally, problems encountered throughout the fieldwork and project as a whole were minor, with the exception of the sustained hard weather. This had a significant effect on the growth of the macrophytes in the new year since the temperatures stayed close to freezing well into March of 2010. Additionally it also created unexpected gaps in the collected data sets, however this does not appear to have caused any negative effects on the results. Once temperatures had begun to rise, the macrophytes grew with their expected vigour and the performance of the systems seemed to have changed little with no dramatic ill effect on nutrient removal. All problems that were encountered throughout the 18-month sampling period were generally corrected within 24 hours of them being recorded.

3. Results

The following results material is based upon samples taken from December 10th, 2008 and June 10th, 2010 (18 months). The sampling was performed weekly and the nutrient analysis performed within 24 hours. The initial 6 months of operation were using as a settling period for all of the treatment systems to mature as there were 2 changes made to the volumetric loading rates. The majority of the results listed here are based upon the data between June 10th, 2009 and June 10th, 2010. The first and last samples were taken on these days exactly, providing a full 12 months of data.

3.1 Initial data

Table 1 provides the summary statistics for the influents of all ICW systems. The influents to the normal, recycling and high flow rate ICW treatment systems are associated with relatively high ammonia-nitrogen (approximately 100 mg/l) and nitrate-nitrogen (about 10 mg/l) concentrations. In comparison, the influent for the high nutrient loading treatment system had ammonia-nitrogen and nitrate-nitrogen concentrations that were considerably higher and lower, respectively. Table 1 indicates also the water quality of the raw liquid fraction of the separated anaerobic digestate, which was diluted with tap water to obtain the desired influents. This was necessary because of the very high concentration of ammonia-nitrogen (roughly between 1600 and 3000 mg/l), which cannot be treated effectively by wetland systems. Moreover, dilution was required to protect *G. maxima* from the toxicity of ammonia. The requirement for pre-treatment of high strength animal wastewater has been reviewed by Cronk (1996) and is a heavily re-occurring feature in many wetland systems dealing with high-ammonia concentration wastewaters. Pre-treatment is not a feature that occurs commonly in ICW systems in Ireland and there are no recorded instances of it being used on a permanent basis in regularly sampled full-scale systems. Some systems that are in operation have a mixing area where yard runoff and wastewaters are collected before discharging to an ICW system, but these are simple collection tanks and not dedicated pre-treatment methods.

A direct comparison of the different treatment systems (Fig. 1) for the two flow phases is difficult due to the variation that the adjusted rates provide. The initial setup of the systems in winter months did mean that their “settling in” and maturing took longer than anticipated. Combining these facts with a dramatic change in the applied influent rate creates a problematic situation in regard to reliable comparisons between the phases. With heavy initial loading during colder seasons with low growth rates and a sudden change in the loading, the effect on the systems would only be reliably recorded if there had been further replication of each system. After each adjustment, there would a lag period in each treatment system and replicate and the systems adjust to the new operational parameters. Even after the 2nd adjustment was made to the systems, a few weeks was given before analysis was started on the June 10th start date for the final 12 months of sampling.

Seasons indicated by temperature fluctuations and different lengths of the two flow phases are likely to have influenced the ICW system performances (Cronk, 1996; Kuschk *et al*., 2003). For example, the first phase (high flow rate) was conducted during winter and early spring, when the ambient temperature was low (<10°c), while the second phase (low flow) was tested starting in late spring and continued on through summer and the remainder of the year with higher ambient temperatures. These contrasting environmental factors make a direct assessment of the absolute water quality values between the 4 observed treatments substantially more difficult. In addition to the difficulties in comparing the results recorded from these systems between their 2 flow rates, any direct comparisons to large-scale wetlands are rather tentative. Similarities can be drawn from the results and conclusions, but substantial differences in construction (materials, liners and operational period) make them difficult. However, considering that all the meso-scale systems were operated with different test conditions in parallel, it is justified to compare system performances relative to each after the initial 6 months of operation, as each system had matured to a level where the macrophytes had rooted well into the substrate medium and had acclimated to the nutrient-rich inflow. As such, the data provided in this section deals primarily with the values recorded between June 10th 2009 and June 10th 2010. The initial data from the first 6 months of operation is used occasionally and noted where it is used. All statistical data is based upon the latter 12 months of sampling. Some seasonal variation and deviation between treatments was recorded with some unexpected results from some of the mechanisms in place, such as recycling (section 4.6) and the prolonged lack of any effluent from the replicates within that treatment.

3.2 Standard treatment

Standard treatment, which is the basis for the other 3 treatments, comprises of common functional parameters from treatment wetlands throughout the world; low ammonia-nitrogen inflow, medium hydraulic loading and no pre-treatment. These systems produced consistently good results for the latter 12 months (low flow), with average removal efficiencies of 99.46%, 97.92%, 96.12%, 65.89%, 74.42% and 21.88% for Ammonia, MRP, Nitrite, Nitrate, TON and Chloride respectively. Similar removal rates were recorded in all 4 treatment methods and indeed within each system replicate itself. The standard treatment takes the basic design concepts that are incorporated into a myriad of constructed wetland designs, including that of the ICW concept and design. The removal rates recorded are on par with that of full-scale systems dealing with agricultural, dairy and municipal wastewaters which treat influents with a similar level of ammonia-N in their influent (Harrington *et al*., 2011, Babatunde *et al*., 2008; Scholz *et al*., 2007). The seasonal variation was generally minimal throughout the full sampling period, though there is a noticeable increase in the colder winter months of December 2009 and January 2010. Whilst the average effluent concentrations did go above the 0.5 mg/l imposed limit, the full averages were generally below this. The drinking water standards that are currently in use in Ireland state that water should not have an ammonium-N concentration greater than 0.3 mg/l (S.I no. 106, 2007). The ammonia-N content of the effluent from all normal treatment systems comes close to reaching this limit, year-round.

3.2 Recycling

The recycling treatment is the method that has the greatest operational difference to the other treatment methods due to the mechanism put in place to recycle the treated water. The recycling was performed by hand, but ideally in a full-scale system where such an approach is wholly impractical, a mechanical, timed pumping system would perform the same role. The recycling fulfils the role of adding pre-treated water where denitrification has already been taking place in the treated wastewater and essentially is adding pre-nitrified water into the beginning of the treatment system and enhancing and promoting denitrification earlier in the system.

The recycling of the wastewater back through the system has a 4-fold effect on the system overall; it increases the overall hydraulic residence time, dilutes in the influent concentration, adds pre-nitrified water to the first cell and reduces the amount of “clean-” or tap-water that would be needed to dilute the influent to a sufficient concentration to be easily dealt with (storage tank dilutions were 16:1 with

16 parts tap water and 1 part piggery wastewater). In a large-scale system, these effects would stand to have a significant effect on the operational costs and effectiveness of an ICW system.

The increased hydraulic residence time was not accurately measured in any of the replicate systems, either in Recycling or in any of the other treatment methods. This was due to a lack of an agreed approach later in the project before the fieldwork was completed and the lack of dedicated training and time available for its completion. A full analysis of the hydraulic retention time had been suggested using Bromide markers and repeated and high quantities of samples being taken throughout the system over the period of a month and this was simply not feasible without having dedicated personnel to oversee and perform this. However, the increased retention was distinctly noticeable with the extended periods of no through-flow in the Recycling systems during the summer months especially, in comparison to the other treatment systems which had through-flow even in summer.

These systems produced consistently good results for the latter 12 months (low flow), with average removal efficiencies of 99.82%, 97.28%, 97.13%, 79.15%, 83.32% and 21.00% for Ammonia, MRP, Nitrite, Nitrate, TON and Chloride respectively. These average removal rates are very similar to that of Standard treatment, but do yield slightly higher removal rates for TON.

3.2 High Nutrient Loading

High nutrient loading (HNL) expands on the standard treatment by simply doubling the influent ammonia-N concentration whilst retaining the other shared parameters. HNL looked at the feasibility of using a higher concentration of influent in order to allow for great volumes of raw material being used in the influent. This would allow for a higher treatment rate of any stored nutrient-rich wastewaters. Retaining the stead retention and lower flow of Standard treatment, the macrophytes that were in the HNL replicated acclimated to the higher influent concentrations and their growth rates improved as a result of the more readily available nutrients. This was observed visually throughout the growing periods of Spring and Summer. In Standard and Recycling systems, the macrophytes had the strongest growth in cells 1 and 2, whereas in HNL the macrophytes grew much denser and larger in cells 1 and 2 and those located in cell 3 were noticeably larger than those in cell 3 in Standard or Recycling. This can be clearly seen in the Figure 16 below.

Fig 16. Image showing enhanced growth rate of macrophytes in HNL (background systems) in comparison to Standard and Recycling systems (foreground)

The HNL systems, similarly to Recycling also had repeated instances of there being zero effluent from 2 of the replicates during some of the summer months. This may be attributed to the increased macrophyte densities and their increased evapotranspiration rates. These systems coped well throughout the 12 and indeed the

18 months, of operation even with the increased ammonia-N concentrations. These systems produced consistently good results for the latter 12 months (low flow), with average removal efficiencies of 99.61%, 98.52%, 98.36%, -76.99%, 63.22% and 24.07% for Ammonia, MRP, Nitrite, Nitrate, TON and Chloride respectively. The massive increase in Nitrate-N in the system is the most obvious difference between this and other treatment systems. The HNL systems show very similar removal rates of both nitrite-N to that of the other 3 treatments, but with decreased removal of TON. The increase in nitrate-N in the effluent points towards the systems not having housed or supported the full cycling of nitrogen, but it does show that they are capable of effectively reducing the ammonia-N levels to background levels.

3.2 High Flow Rate

The High flow rate (HFR) systems, similarly to HNL, take the baseline design and operational parameters of the Standard treatment and effectively change only one variable to double that of the baseline design, in this instance influent flow rate. Between the two flow rates that were used in Moorepark, the volume of influent entering the HFR systems was substantial in comparison to that of the other treatment systems. During the high-flow period, the HFR replicates were receiving 150m^3/ha/d equivalent, this was dropped to 74m3/ha/d in the low-flow period. This higher influent loading was used as an indication of a full-scale system where a large quantity of material was being produced every day and as a result of that, a higher influent loading would be preferable than a higher concentration or recycling approach. This higher flow rate would allow for more material to be moved and treated in the same period of time as the other approaches.

The HFR systems, unlike Recycling and HNL, never experienced any periods of short, or indeed prolonged periods of zero discharge from the final cell due to the increased volumes being handled. Similarly to that of HNL, the additional nutrients that were being received in the lower cells of the replicates, were having a positive effect on the macrophyte grow rates and densities during the growing period. Having a higher velocity through the system however, did have some marginal effects on the removal efficiencies recorded.

HFR consistently had the highest average levels of ammonia-N in the effluent samples in the latter 12 month period (Fig 16). These regularly were over the applied threshold of 0.5 mg/l. The MRP levels were similar to that of the other treatment systems and follow very closely (Fig 17), though this is consistent in all treatment methods due to the low levels of MRP in the influent since the majority of the insoluble P was removed in the solid fraction. Where HFR does differ is that of the mass nitrogen removal rates (Fig 18), where the average removal rates, when looked at in a mass removal perspective, show a dramatically different result to that of the other systems. The HFR treatment shows that it could theoretically, while operating at the 74m3/ha/d flow rate could remove up to 250kg N per annum. Despite having higher effluent concentrations of ammonia-N, the capacity to remove such mass removals of N would be of a very definite boon to farmers and piggeries alike under the premise of the Nitrates Directive.The HFR systems produced consistently good results for the latter 12 months (low flow), with average removal efficiencies of 98.02%, 97.16%, 77.54%, -23.80%, 4.53% and 6.74% for Ammonia, MRP, Nitrite, Nitrate, TON and Chloride respectively. The distinct differences with HFR are the lower Nitrite removal levels along with the increased nitrate levels. This is similar to what is observed in the HNL systems, but less dramatic. However, the very low levels of TON and Chloride removal is noticeable and may well be a direct result of the faster throughput of the HFR systems.

Table 2. Average nutrient values for raw, storage tank 1 and storage tank 2

Variable	Unit	High Flow Rate[c]			Low Flow Rate[d]			12 month range		
		Mean	SD[e]	n[f]	Mean	SD	n	mean	SD	n
Raw liquid fraction of the separated anaerobic digestate used as a source material for storage tanks 1 and 2										
TON[a]	*mg/l*	1.61	n/a[g]	1	0.08	0.154	19	0.08	0.154	19
Ammonia-nitrogen	*mg/l*	1629.19	n/a[g]	1	3036.02	802.765	18	3036.02	802.765	18
Nitrite-nitrogen	*mg/l*	0.01	n/a[g]	1	0.12	0.296	20	0.12	0.296	20
Nitrate-nitrogen	*mg/l*	1.67	n/a[g]	1	0.07	0.164	20	0.07	0.164	20
MRP[b]	*mg/l*	15.63	n/a[g]	1	51.79	29.076	19	51.79	29.076	19
Chloride	*mmol/l*	651.5	n/a[g]	1	1216.5	191.47	20	1216.5	191.47	20
Liquid from storage tank 1, providing influent to the normal, recycling and high flow rate treatments systems (100mg/l ammonia-nitrogen)										
TON	*mg/l*	13.48	4.46	2	9.29	3.899	22	9.29	3.899	22
Ammonia-nitrogen	*mg/l*	97.73	8.874	2	98.14	20.936	20	98.14	20.936	20
Nitrite-nitrogen	*mg/l*	0.15	0.134	2	2.05	2.12	22	2.05	2.120	22
Nitrate-nitrogen	*mg/l*	13.33	4.327	2	6.93	3.854	23	6.93	3.854	23
MRP	*mg/l*	1.12	0.157	2	1.6	0.361	23	1.60	0.361	23
Chloride	*mmol/l*	60.7	7.55	2	55.1	7.84	23	55.1	7.84	23
Liquid from storage tank 2, providing influent to the high nutrient loading treatment system (200 mg/l ammonia-nitrogen)										
TON	*mg/l*	7.16	8.825	2	13.07	7.799	22	13.07	7.799	22
Ammonia-nitrogen	*mg/l*	145.43	6.914	2	179.34	40.708	20	179.34	40.708	20
Nitrite-nitrogen	*mg/l*	0.81	1.108	2	9.55	6.82	22	9.55	6.820	22
Nitrate-nitrogen	*mg/l*	6.36	7.717	2	2.75	2.529	23	2.75	2.529	23
MRP	*mg/l*	1.72	0.153	2	3.25	0.599	23	3.25	0.599	23
Chloride	*mmol/l*	85.6	6.62	2	96.8	11.75	23	96.8	11.75	23

Note: The biochemical oxygen demand was not measured for the inflow. [a] Total organic nitrogen. [b] Molybdate reactive phosphorus. [c] 112 and 180 m^3/ha/d for the normal and elevated flow operational modes, respectively. [d] 37 and 74 m^3/ha/d for the normal and elevated flow operational modes, respectively. [e] Standard deviation. [f] Sample number. [g] Not applicable Values given for "Low Flow Rate" and "12 month range"are identical as they cover the same period of the project.

Table 3. Average nutrient values for outflows from cell 1 and cell 4 in Normal treatment

Variable	Unit	High flow rate			Low flow rate			12 month range		
		Mean	SD	n	mean	SD	n	mean	SD	n
Outflow of wetland cell 1										
BOD	*mg/l*	19.8	3.44	7	48.4	22.61	27	50.8	22.08	23
TON	*mg/l*	0.97	1.106	15	7.62	9.888	53	8.55	10.512	44
Ammonia-nitrogen	*mg/l*	63.75	16.194	15	18.33	17.349	52	14.34	15.360	43
Nitrite -nitrogen	*mg/l*	0.31	0.469	15	0.64	0.541	53	0.69	0.535	44
nitrate-nitrogen	*mg/l*	0.67	0.813	15	6.97	9.843	54	7.83	10.482	45
MRPc	*mg/l*	0.27	0.106	15	0.15	0.106	53	0.16	0.115	44
Chloride	*mmol/l*	47.6	3.99	15	55.2	27.16	54	59.9	27.14	45
pH	-	7.84	0.433	12	7.2	0.178	26			
Dissolved oxygen	*mg/l*	3.6	1.07	10	3.0	1.68	26			
Temperature	^oC	7.7	2.75	12	14.5	3.74	26			
Conductivity	*µS*	1488	219.5	12	1341	242.7	26			
Outflow of wetland cell 4										
BOD	*mg/l*	13.9	9.58	7	18.7	15.22	27	18.0	15.75	23
TON	*mg/l*	0.1	0.196	15	2.28	4.077	53	2.38	4.342	44
Ammonia-nitrogen	*mg/l*	16.43	17.107	15	2.31	5.696	53	0.53	0.610	44
Nitrite -nitrogen	*mg/l*	0.02	0.017	15	0.11	0.222	54	0.08	0.178	45
nitrate-nitrogen	*mg/l*	0.09	0.195	15	2.21	4.042	54	2.37	4.305	45
MRPc	*mg/l*	0.04	0.032	15	0.04	0.072	53	0.03	0.078	44
Chloride	*mmol/l*	28.5	14.08	15	41.4	14.31	54	43.1	14.72	45
pH	-	7.85	0.655	12	7.35	0.160	26			
Dissolved oxygen	*mg/l*	5.2	1.48	10	4.2	1.82	26			
Temperature	^oC	7.7	2.96	12	14.6	3.76	26			
Conductivity	*µS*	817	303.6	12	604	128.9	26			

Note: The temperature was measured in the laboratory. [a] Five-days at 20 °C N-allythiourea biological oxygen demand. [b] Total organic nitrogen. [c] Molybdate reactive phosphorus. [d] 112 and 180 m^3/ha/d for the normal and elevated flow operational modes, respectively. [e] 37 and 74 m^3/ha/d for the normal and elevated flow operational modes, respectively. [f] Standard deviation. [g] Sample number.

Table 4. Average nutrient values for the outflows from cell 1 and cell 4 in Recycling treatment

Variable	Unit	High flow rate			Low flow rate			12 month range		
		Mean	SDf	n	mean	SD	n	mean	SD	n
Outflow of wetland cell 1										
BOD	*mg/l*	19.4	6.09	7	40.1	23.46	27	41.3	24.68	23
TON	*mg/l*	0.97	1.200	15	5.17	6.977	53	6.03	7.365	44
Ammonia-nitrogen	*mg/l*	61.54	14.921	15	13.56	14.348	52	10.86	13.100	43
Nitrite -nitrogen	*mg/l*	0.39	0.732	15	0.58	1.43	54	0.66	1.554	45
nitrate-nitrogen	*mg/l*	0.60	0.885	15	4.7	6.856	54	5.49	7.264	45
MRPc	*mg/l*	0.35	0.140	15	0.18	0.148	53	0.17	0.160	44
Chloride	*mmol/l*	49.2	4.00	15	69.3	42.92	54	77.8	41.84	45
pH	-	7.64	0.363	12	7.22	0.142	26			
Dissolved oxygen	*mg/l*	4.2	2.94	10	2.9	1.42	26			
Temperature	^oC	7.7	2.71	12	14.5	3.49	26			
Conductivity	*µS*	1499	181.5	12	1313	342.1	26			
Outflow of wetland cell 4										
BOD	*mg/l*	12.3	8.47	7	16.1	18.8	27	15.1	19.61	23
TON	*mg/l*	0.45	1.077	15	1.49	2.191	53	1.55	2.378	44
Ammonia-nitrogen	*mg/l*	13.57	13.491	15	1.09	3.324	53	0.17	0.120	44
Nitrite -nitrogen	*mg/l*	0.03	0.023	15	0.08	0.226	54	0.06	0.204	45
nitrate-nitrogen	*mg/l*	0.44	1.075	15	1.37	2.181	54	1.45	2.367	45
MRPc	*mg/l*	0.12	0.150	15	0.06	0.083	53	0.05	0.087	44
Chloride	*mmol/l*	31.2	12.27	15	41.9	19.48	54	43.6	20.68	45
pH	-	7.87	0.643	12	7.41	0.144	26			
Dissolved oxygen	*mg/l*	6.7	1.80	10	4.2	1.60	26			
Temperature	^oC	7.8	2.08	12	14.7	3.48	26			
Conductivity	*µS*	814	296.1	12	635	110.2	26			

Note: The temperature was measured in the laboratory. [a] Five-days at 20 °C N-allythiourea biological oxygen demand. [b] Total organic nitrogen. [c] Molybdate reactive phosphorus. [d] 112 and 180 m^3/ha/d for the normal and elevated flow operational modes, respectively. [e] 37 and 74 m^3/ha/d for the normal and elevated flow operational modes, respectively. [f] Standard deviation. [g] Sample number.

Table 5. Average nutrient values for effluent from cell 1 and cell 4 from High Nutrient Loading treatment

Variable	Unit	High flow rate			Low flow rate			12 month range		
		Mean	SDf	n	mean	SD	n	mean	SD	n
Outflow of wetland cell 1										
BOD	*mg/l*	41.5	12.66	7	71.4	42.83	27	74.4	45.77	23
TON	*mg/l*	0.87	1.398	15	10.19	11.559	53	12.10	11.810	44
Ammonia-nitrogen	*mg/l*	124.46	24.143	15	58.96	39.852	52	48.71	35.990	43
Nitrite -nitrogen	*mg/l*	0.06	0.100	15	0.86	1.083	54	1.01	1.126	45
nitrate-nitrogen	*mg/l*	0.81	1.351	15	10	11.98	54	11.84	12.321	45
MRPc	*mg/l*	1.26	0.285	15	1.26	2.913	53	1.39	3.189	44
Chloride	*mmol/l*	85.2	10.35	15	71.4	27.85	54	72.3	30.15	45
pH	-	7.9	0.243	12	7.44	0.24	26			
Dissolved oxygen	*mg/l*	2.4	2.49	10	1.9	1.05	26			
Temperature	^oC	7.8	2.66	12	14.8	3.21	26			
Conductivity	*µS*	2303	306.5	12	2109	282.3	26			
Outflow of wetland cell 4										
BOD	*mg/l*	18.8	12.09	7	19.7	11.56	27	17.3	9.90	23
TON	*mg/l*	0.46	0.858	15	4.20	7.681	53	4.81	8.297	44
Ammonia-nitrogen	*mg/l*	43	41.655	15	3.62	9.184	53	0.69	1.310	44
Nitrite -nitrogen	*mg/l*	0.02	0.017	15	0.18	0.288	54	0.16	0.285	45
nitrate-nitrogen	*mg/l*	0.45	0.857	15	4.22	7.667	54	4.87	8.246	45
MRPc	*mg/l*	0.18	0.199	15	0.23	0.838	53	0.26	0.919	44
Chloride	*mmol/l*	51.6	23.74	15	70.7	30.89	54	73.5	32.75	45
pH	-	7.95	0.437	12	7.38	0.164	26			
Dissolved oxygen	*mg/l*	4.7	2.32	10	3.3	1.50	26			
Temperature	^oC	7.8	2.77	12	14.8	3.28	26			
Conductivity	*µS*	1252	643.6	12	842	220.3	26			

Note: The temperature was measured in the laboratory. [a] Five-days at 20 °C N-allythiourea biological oxygen demand. [b] Total organic nitrogen. [c] Molybdate reactive phosphorus. [d] 112 and 180 m3/ha/d for the normal and elevated flow operational modes, respectively. [e] 37 and 74 m^3/ha/d for the normal and elevated flow operational modes, respectively. [f] Standard deviation. [g] Sample number.

Table 6. Average nutrient values for effluent from cell 1 and cell 4 from High Flow Rate treatment

Variable	**Unit**	**High flow rate**			**Low flow rate**			**12 month range**		
		Mean	**SDf**	**n**	**mean**	**SD**	**n**	**mean**	**SD**	**n**
Outflow of wetland cell 1										
BOD	*mg/l*	31.2	12.93	7	89.4	60.72	27	92.7	60.20	23
TON	*mg/l*	1.84	2.193	15	7.57	8.979	53	8.93	9.282	44
Ammonia-nitrogen	*mg/l*	79.51	13.754	15	35.28	17.727	52	31.49	16.790	43
Nitrite -nitrogen	*mg/l*	0.46	0.794	15	0.77	1.147	54	0.86	1.225	45
nitrate-nitrogen	*mg/l*	1.39	1.785	15	6.92	8.968	54	8.18	9.328	45
MRPc	*mg/l*	0.64	0.247	15	1.07	3.057	53	1.24	3.334	44
Chloride	*mmol/l*	60	13.57	15	49.1	21.58	54	52.1	22.00	45
pH	-	7.65	0.291	12	7.22	0.175	26			
Dissolved oxygen	*mg/l*	3.1	1.82	10	1.8	1.21	26			
Temperature	^oC	8	2.65	12	15.1	3.14	26			
Conductivity	*µS*	1753	221.9	12	1606	223.7	26			
Outflow of wetland cell 4										
BOD	*mg/l*	13.7	8.76	7	15.8	9.16	27	14.2	8.32	23
TON	*mg/l*	0.48	0.949	15	7.52	13.129	53	8.87	14.037	44
Ammonia-nitrogen	*mg/l*	23.79	22.003	15	4.64	7.602	53	2.07	3.580	44
Nitrite -nitrogen	*mg/l*	0.03	0.022	15	0.42	1.713	54	0.46	1.871	45
nitrate-nitrogen	*mg/l*	0.47	0.941	15	7.27	12.914	54	8.58	13.787	45
MRPc	*mg/l*	0.05	0.025	15	0.24	0.868	53	0.27	0.952	44
Chloride	*mmol/l*	36.4	11.82	15	48.3	16.31	54	51.4	15.63	45
pH	-	7.68	0.369	12	7.33	0.139	26			
Dissolved oxygen	*mg/l*	5.1	1.93	10	3	1.47	26			
Temperature	^oC	8	2.70	12	15.1	3.16	26			
Conductivity	*µS*	968	379.9	12	839	133.2	26			

Note: The temperature was measured in the laboratory. [a] Five-days at 20 °C N-allythiourea biological oxygen demand. [b] Total organic nitrogen. [c] Molybdate reactive phosphorus. [d] 112 and 180 m^3/ha/d for the normal and elevated flow operational modes, respectively. [e] 37 and 74 m^3/ha/d for the normal and elevated flow operational modes, respectively. [f] Standard deviation. [g] Sample number.

Table 7. Average percentage removal rates

Parameter	Normal	Recycle	HNL	HFR
Ammonia	99.46	99.82	99.61	98.02
MRP	97.92	97.28	98.52	97.16
Nitrite	96.12	97.13	98.36	77.54
Nitrate	65.89	79.15	-76.99	-23.80
TON	74.42	83.32	63.22	4.53
Chloride	21.88	21.00	24.07	6.74

Note: minus values indicate a percentage increase in concentration (e.g. - Nitrate in HNL)

Table 8. Average seasonal nutrient percentage removal (June 2009 – June 2010)

	Treatment	**Normal**	**Recycle**	**HNL**	**HFR**
Ammonia	Summer	99.87	99.88	99.81	99.64
	Autumn	99.88	99.86	99.91	99.51
	Winter	98.67	99.77	99.41	96.01
	Spring	99.27	99.79	99.29	96.56
MRP	Summer	97.66	95.85	97.73	96.70
	Autumn	96.98	96.82	98.32	96.39
	Winter	98.95	98.62	99.34	98.04
	Spring	98.38	97.98	98.77	97.71
Nitrite	Summer	98.28	98.49	98.93	95.96
	Autumn	99.07	98.65	99.24	95.31
	Winter	87.87	91.14	96.97	19.84
	Spring	97.16	98.70	98.01	85.58
Nitrate	Summer	90.72	88.55	61.27	90.65
	Autumn	96.78	95.86	87.01	90.41
	Winter	86.97	86.96	47.56	-31.03
	Spring	1.12	49.72	-432.82	-221.24
TON	Summer	92.72	91.12	91.07	92.14
	Autumn	97.39	96.61	96.71	91.81
	Winter	87.58	88.30	86.74	-15.46
	Spring	24.27	58.60	-13.71	-148.18
Chloride	Summer	30.42	27.84	19.48	34.57
	Autumn	31.75	46.58	40.53	4.65
	Winter	40.33	32.94	50.24	14.02
	Spring	-7.33	-18.10	-6.99	-17.61

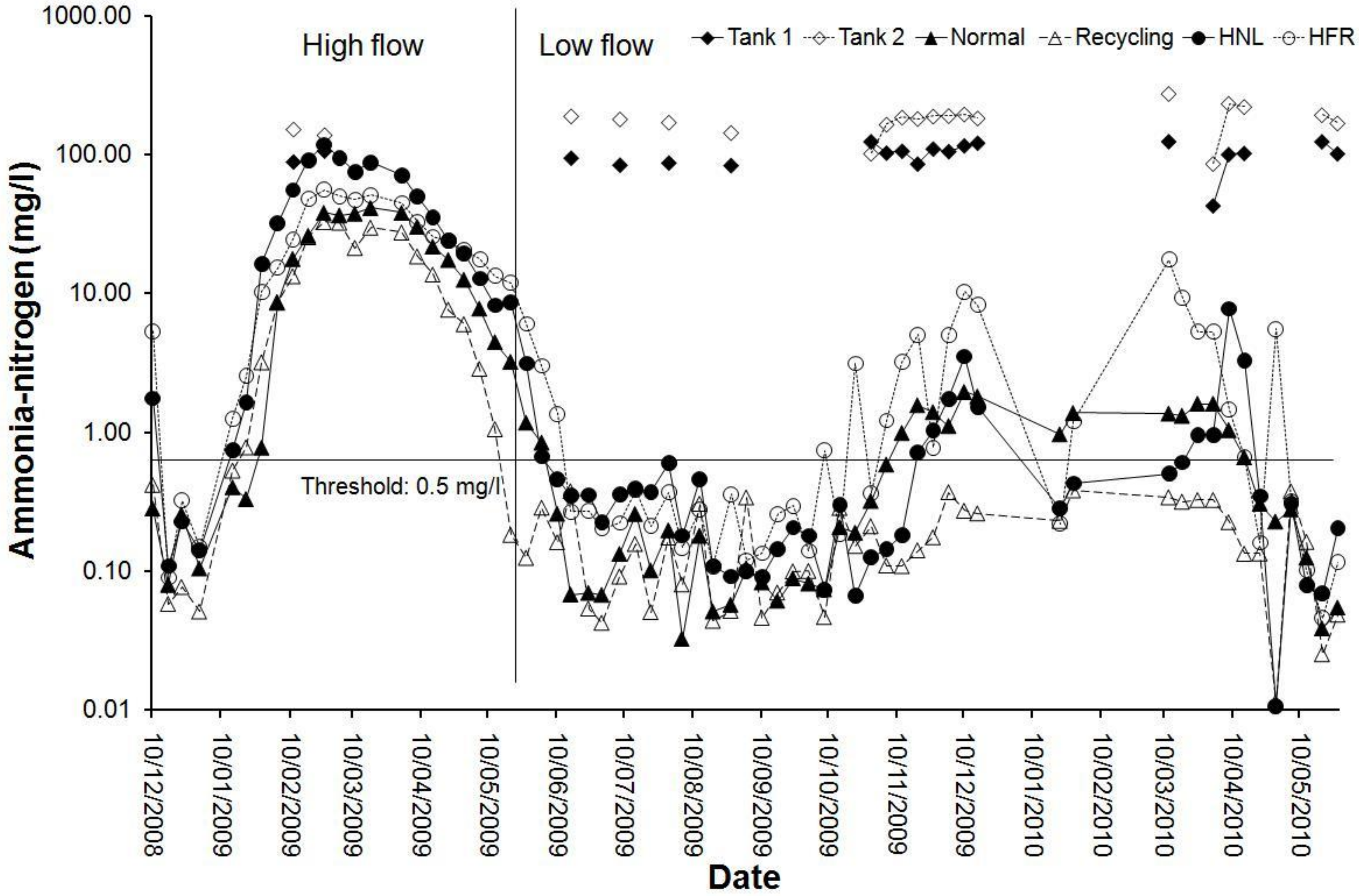

Figure 16. Average ammonia-nitrogen concentration in outflow of cell 4 from each treatment over complete sampling period

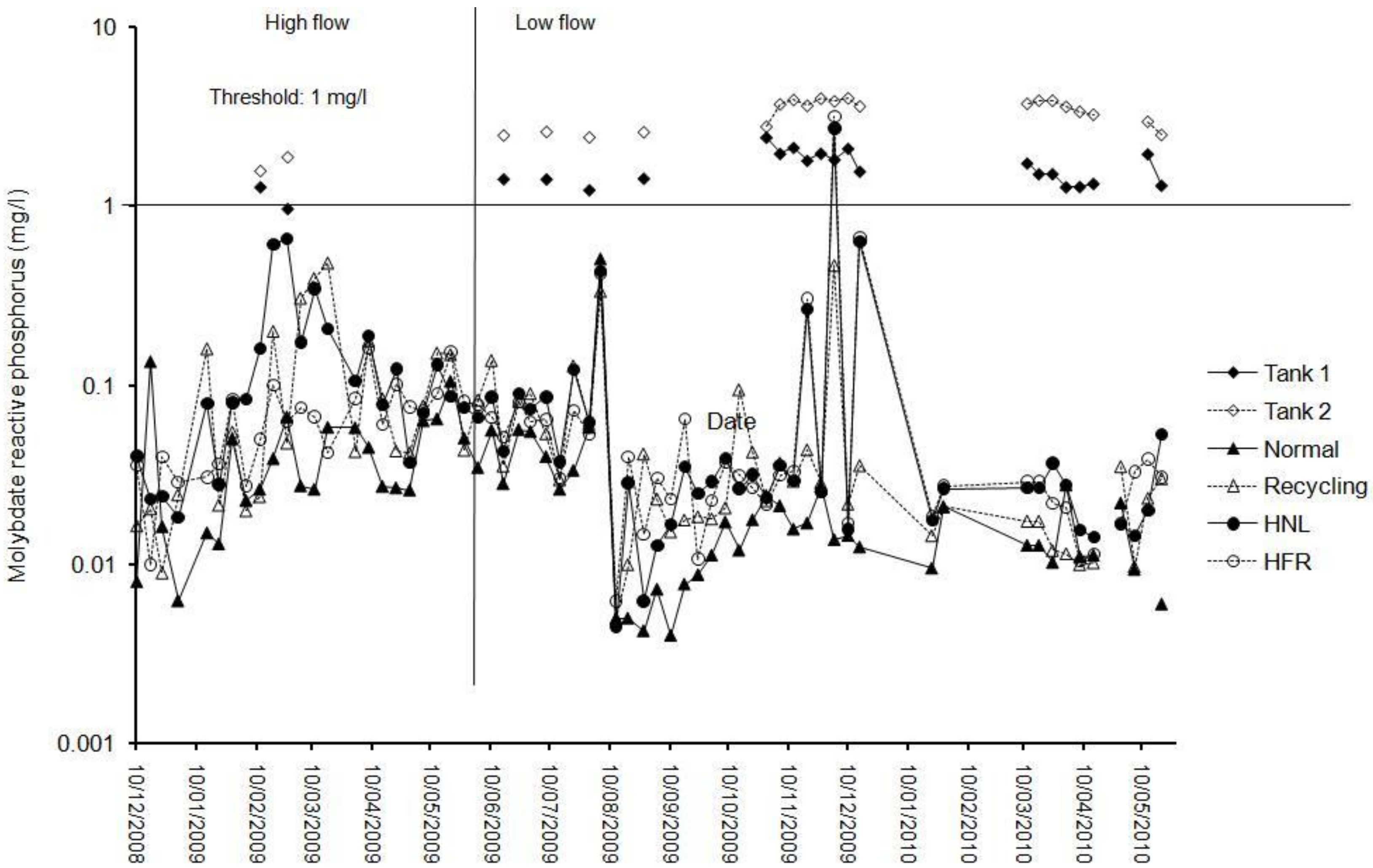

Figure 17. Average MRP concentration from outflow of cell 4 of each treatment over complete sampling period

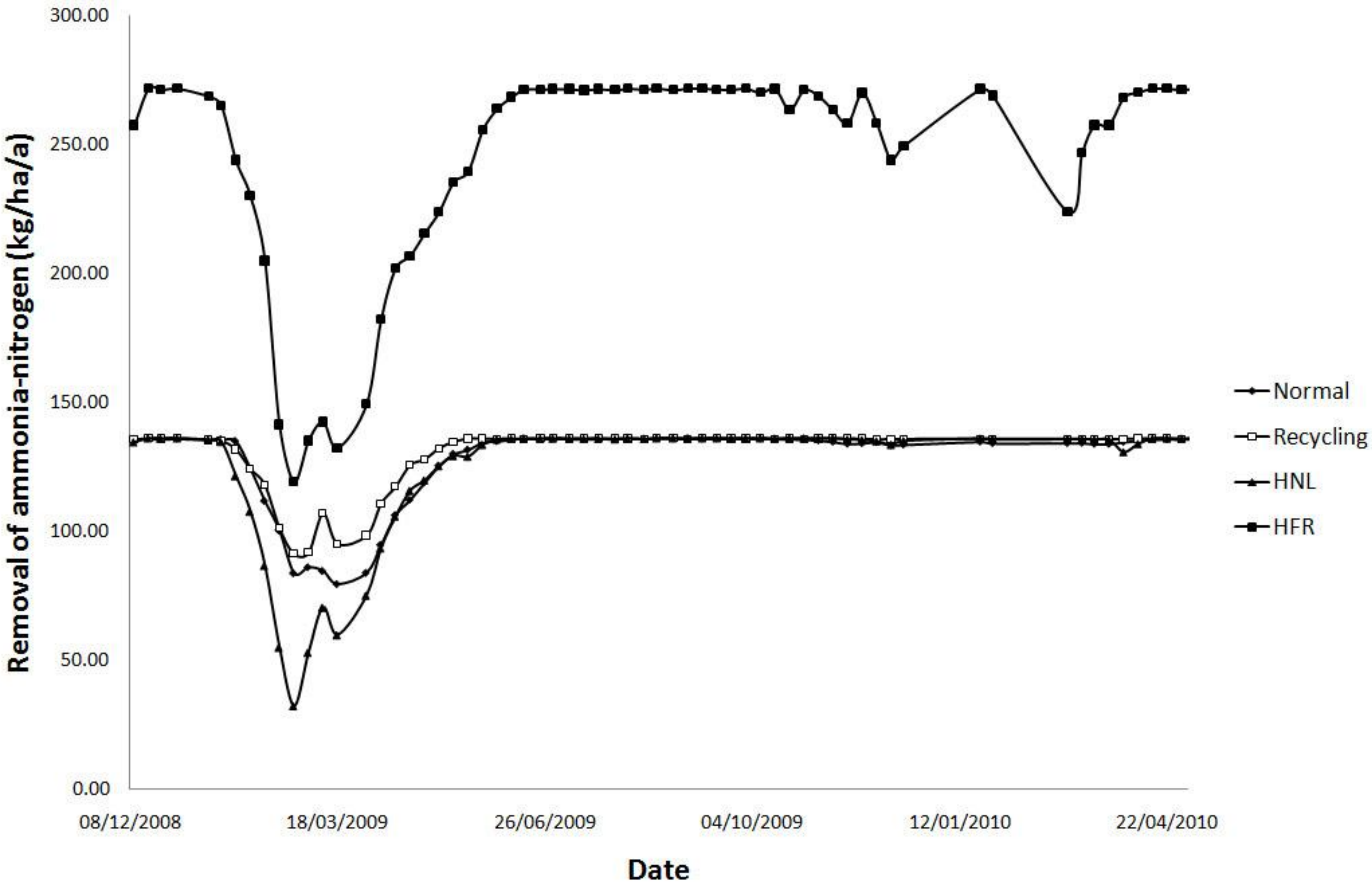

Figure 18. Extrapolated total nitrogen removal (kg/ha/a) from each system. Data is based upon extrapolated tank data and average removal efficiencies of each treatment method

3.3 Statistics:

Statistical tests were carried out using SPSS version 18 (SPSS Inc., Chicago) and all tests are two-tailed unless otherwise stated.

Kolmorgorov-Smirnov test is applied to sets of data in order to ascertain if the samples come from the same (normal) distribution. In the case of the meso-scale sampling method, there were 4 samples in each data set. The test was used to determine if the samples from this study were normally distributed, however the majority of the samples were not normally distributed.

Since the majority of the data were not normally distributed, a non-parametric test, Independent Samples Kruskal-Wallis Test, was applied to the data to determine significant differences. This test is generally applied to compare three or more samples and tests the null hypothesis for all data.

Finally, after the independent sample tests had been concluded, the data was then examined in further detail using Post-Hoc (Mann-Whitney U) tests to examine the significant differences and a bonferroni correction was applied to account for the multiple tests Field, 2005). The Mann-Whitney U test is known to be more efficient when dealing with non-normal distributions in comparison to the more commonly used t-test. The Bonferroni correction is a correction that is used when several dependent or independent statistical tests are being run at the same time. It is applied in order to remove the likelihood of a lot of false positives. The data /values are given an "*a*" value and these are cross-examined at a significance level of *a/n*. The values that are given from the cross-examination of variance in the samples gives U value. These are noted in the figures below.

The figures in sections 3.3.1 - 3.4.1 are graphical representations of the statistical analysis performed on their respective datasets where there was a significant difference between replicates and other systems. For instance, figure 19 shows the replicates from HFR, which yielded significantly difference U values from other treatments for the outflow of cell 1. The first replicate within that treatment was also significantly different to the fourth replicate.

These significant differences between treatments/replicates are more common in Cell 1 outflow analysis than in Cell 4 analysis due to the more variable nature of the influent material entering Cell 1 of each treatment and the replicates therein.

Direct comparisons between each individual replicates with every other replicate was proposed upon the commencing the statistical analysis, however this was deemed to be impractical and problematic. Additionally, the lack of cell 2 & 3 data is a shortcoming to examine where in the system the differing treatments reach a level of similarity such that the effluent is not significantly different from one another.

Statistical analysis was done in conjunction with Dr. Favel Naulty from the Biology Department in University College Dublin. Dr. Naulty peformed the analysis, as described above after discussions with the author and their supervisor regarding the examinations that would be required for the data. All data sets had been tripled checked prior to statistical analysis and any value entries in the data set that were below levels of detection, had their value set at the limit of detection for the Konelab20 (Thermo Clinical Labsystems) which was the principle equipment that was used in the nutrient analysis of all samples taken.

Sections 3.3.1 and 3.3.2 of the manuscript show figures where significant results, as highlighted in the preceding tables, are recorded. The values given in each figure are;

- df - Degrees of freedom.
- H - test statistic for the Kruskal Wallis (H) test.
- U - calculated value from Mann-Whitney U test.

3.3.1 Cell1 outflow analysis

Table 9. Cell 1 outflow analysis
Note: Significant tests are highlighted.

	Normal	**Recycle**	**HNF**	**HFR**
Ammonia	H: 7.266 df: 3 p = 0.064, ns	H: 4.614 df: 3 p = 0.202, ns	H: 1.713 df: 3 p = 0.634, ns	H: 8.099 df: 3 p = 0.044
Orthophosphate	H: 22.538 df: 3 p = 0.000	H: 6.915 df: 3 p = 0.075, ns	H: 2.810 df: 3 p = 0.422, ns	H: 5.788 df: 3 p = 0.122, ns
Nitrite	H: 5.835 df: 3 p = 0.120, ns	H: 7.532 df: 3 p = 0.057, ns	H: 0.958 df: 3 p = 0.811, ns	H: 4.609 df: 3 p = 0.203, ns
Nitrate	H: 2.894 df: 3 p = 0.408, ns	H: 3.052 df: 3 p = 0.384, ns	H: 2.798 df: 3 p = 0.424, ns	H: 1.073 df: 3 p = 0.784, ns
TON	H: 2.852 df: 3 p = 0.415, ns	H: 4.282 df: 3 p = 0.233, ns	H: 4.390 df: 3 p = 0.222, ns	H: 1.719 df: 3 p = 0.633, ns
Chloride	H: 5.401 df: 3 p = 0.145, ns	H: 6.652 df: 3 p = 0.084, ns	H: 19.598 df: 3 p = 0.000	H: 7.873 df: 3 p = 0.049
BOD	H: 11.228 df: 3 p = 0.011	H: 8.340 df: 3 p = 0.039	H: 3.923 df: 3 p = 0.270, ns	H: 0.562 df: 3 p = 0.905, ns
PH	H: 9.973 df: 3 p = 0.019	H: 3.538 df: 3 p = 0.316, ns	H: 1.430 df: 3 p = 0.699, ns	H: 0.416 df: 3 p = 0.937, ns
Conductivity	H: 3.743 df: 3 p = 0.291, ns	H: 2.567 df: 3 p = 0.463, ns	H: 2.580 df: 3 p = 0.461, ns	H: 0.911 df: 3 p = 0.823, ns
DO	H: 2.174 df: 3 p = 0.537, ns	H: 0.432 df: 3 p = 0.934, ns	H: 1.992 df: 3 p = 0.574, ns	H: 1.558 df: 3 p = 0.669, ns
Temperature	H: 0.367 df: 3 p = 0.947, ns	H: 0.079 df: 3 p = 0.994, ns	H: 0.117 df: 3 p = 0.990, ns	H: 1.068 df: 3 p = 0.785, ns

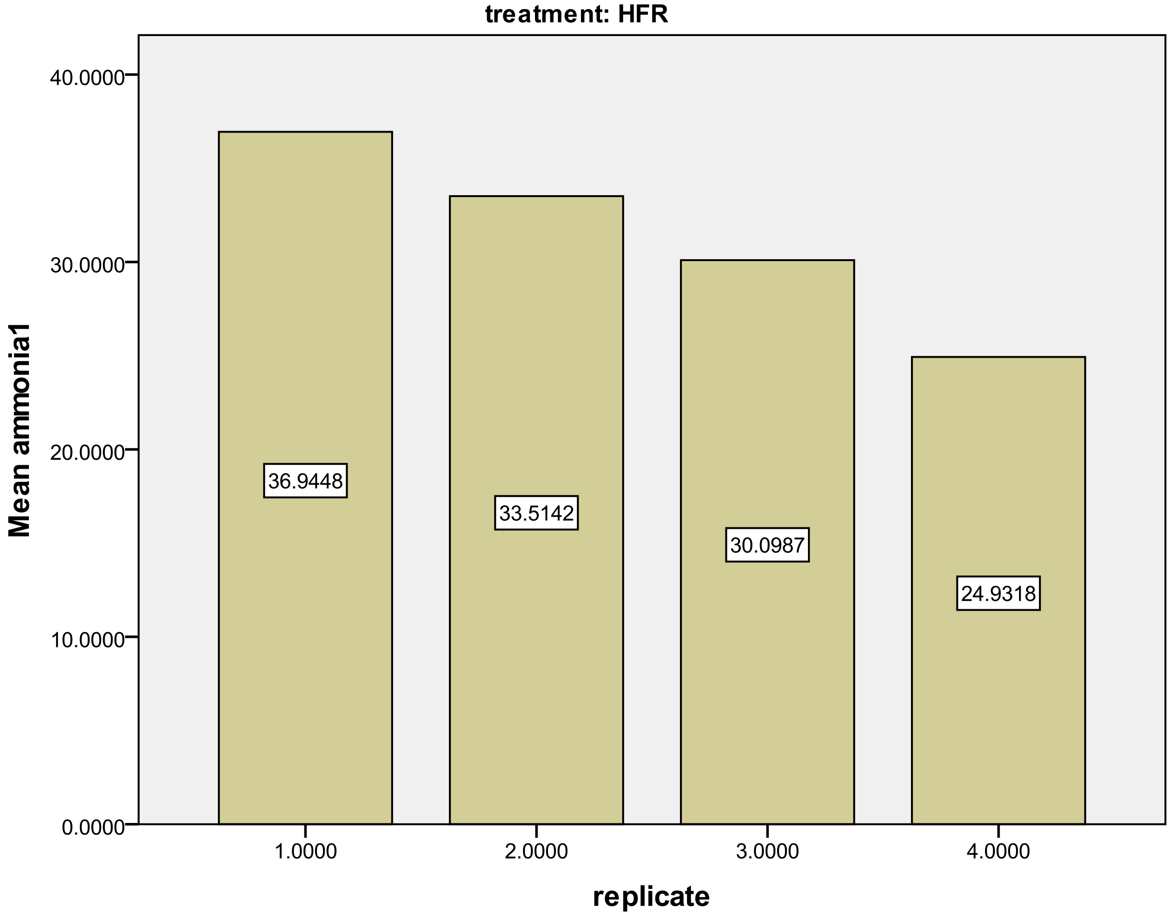

Figure 19. Replicate 1 is significantly different to replicate 4 (U = 623).

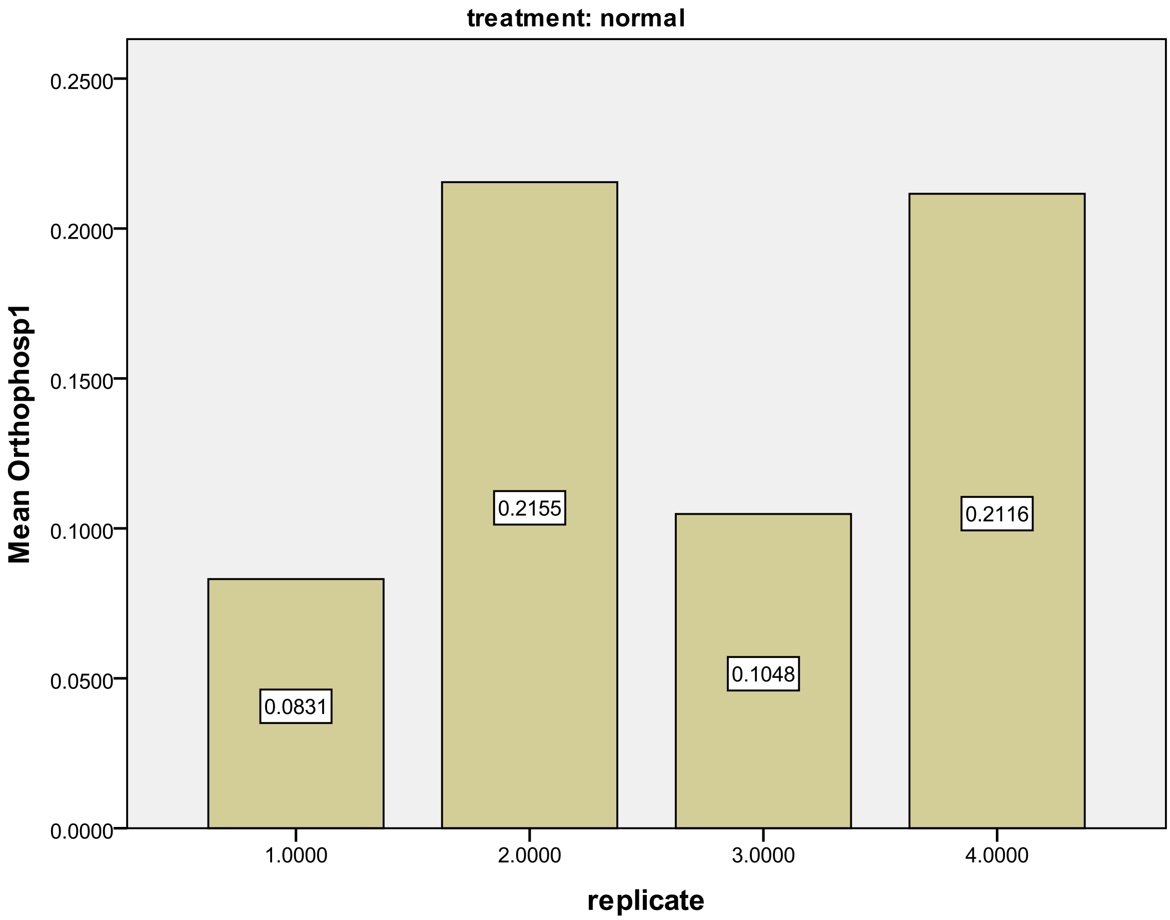

Figure 20. Replicate 1 is significantly different to replicate 2 (U = 545.5) and replicate 4 (U = 505.5). Replicate 3 is significantly different to replicate 4 (U = 647.5)

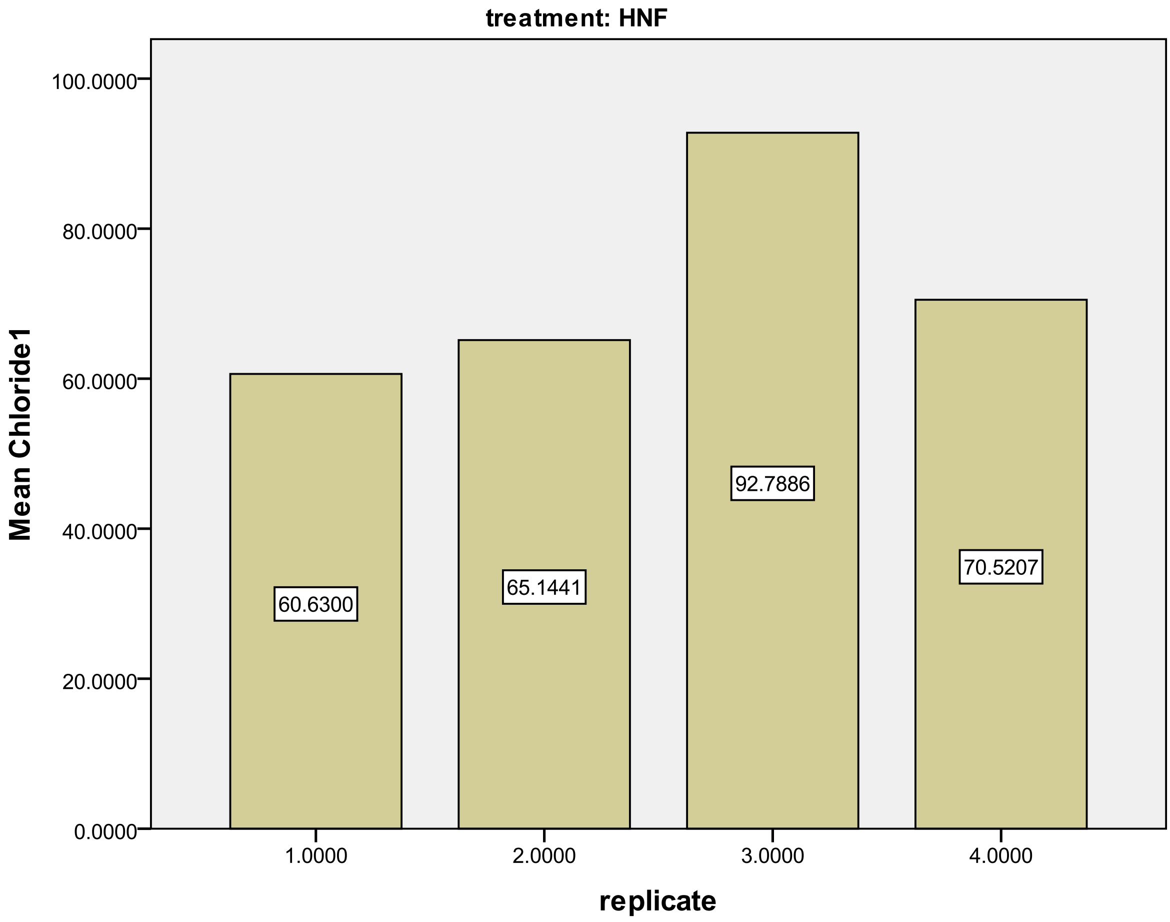

Figure 21. Replicate 3 is significantly different to replicate 1 (U = 488) and replicate 2 (U = 548.5).

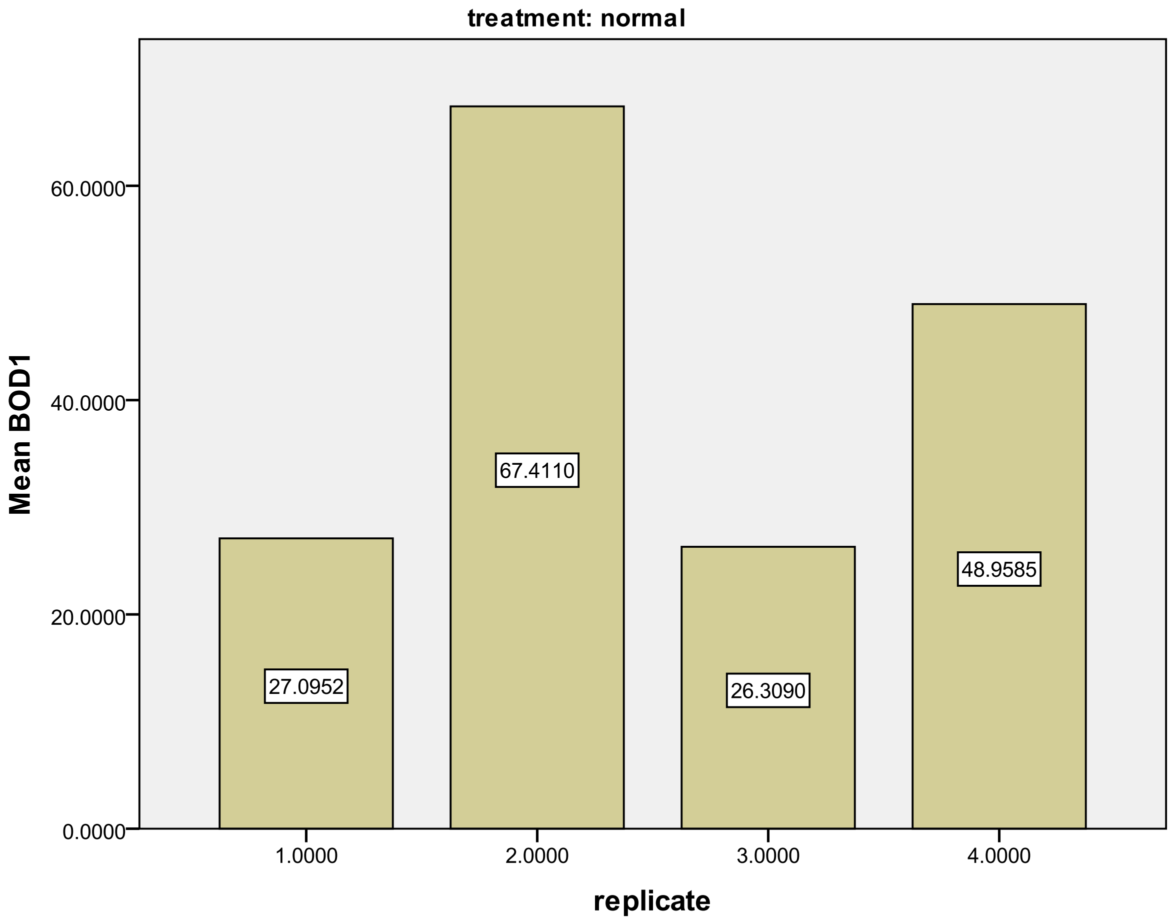

Figure 22. Replicate 2 is significantly different to replicate 1 (U = 30) and replicate 3 (U = 36).

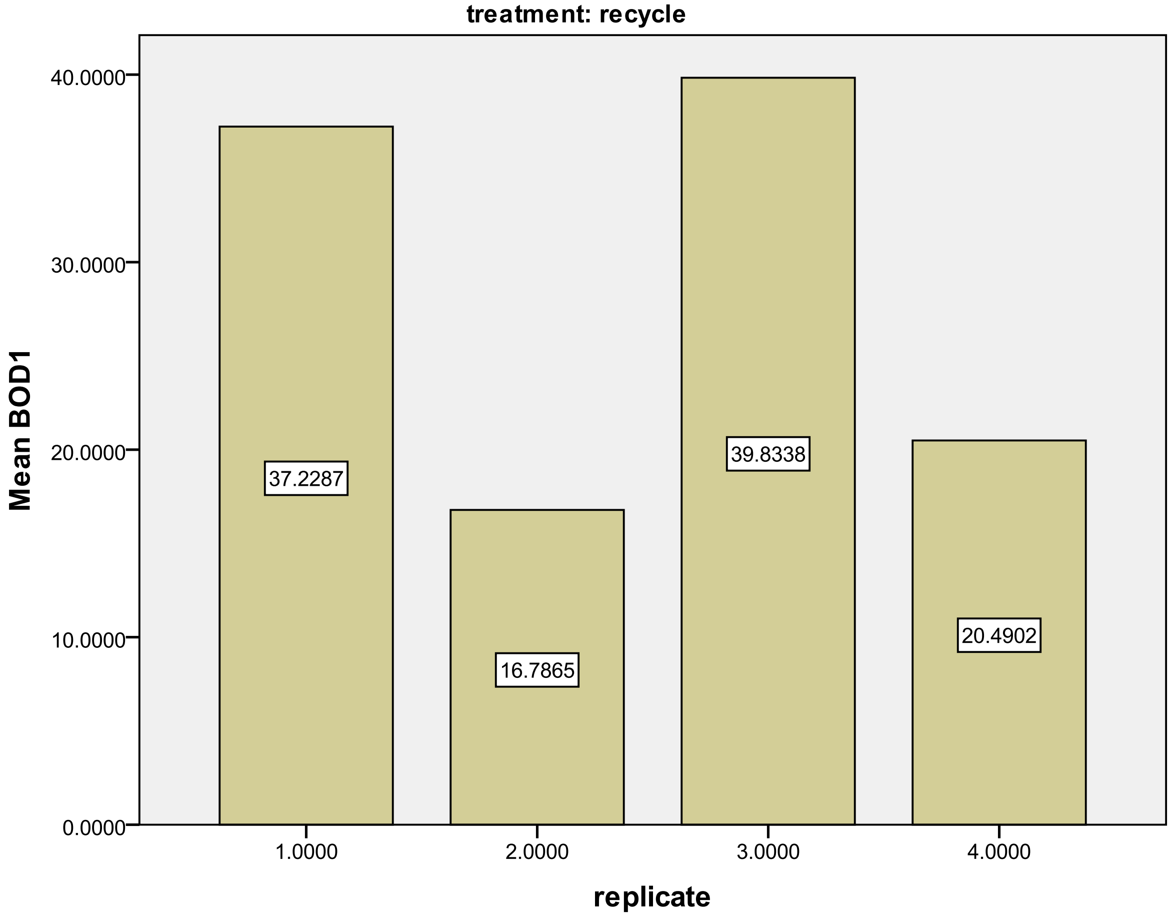

Figure 23. Replicate 1 is significantly different to replicate 2 (U = 33).

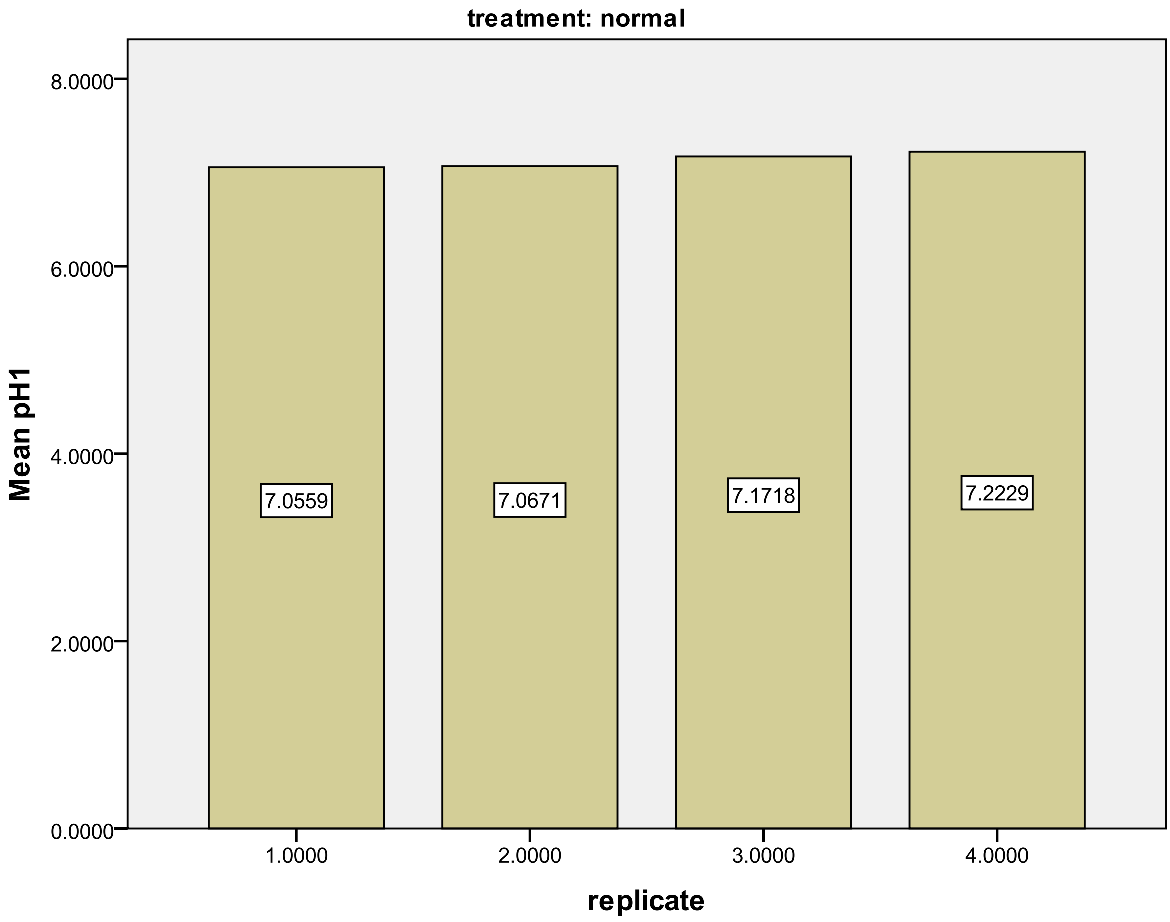

Figure 24. Replicate 4 is significantly different to replicate 1 (U = 74.5) and replicate 2 (U = 72).

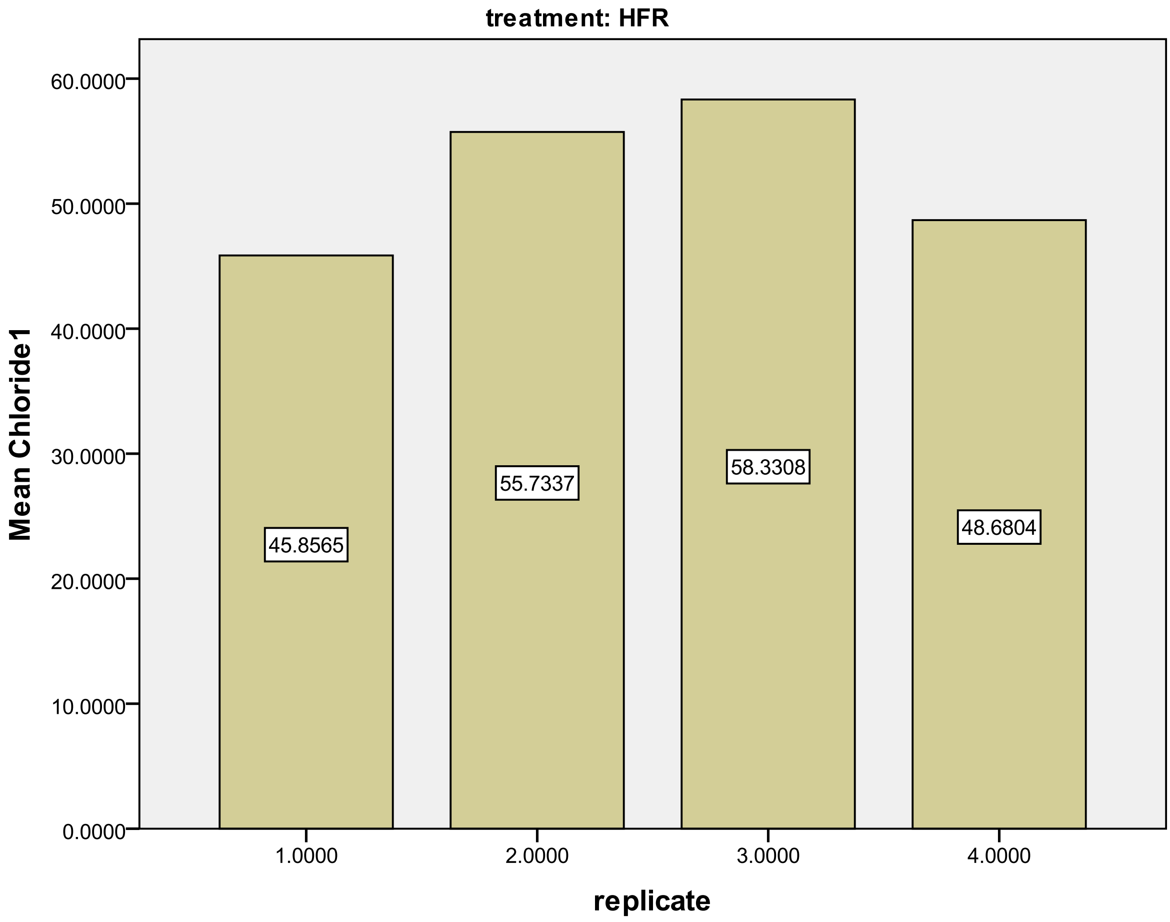

Figure 25. Replicate 3 is significantly different to replicate 1 (U = 488), 2 (U = 548.5) and 4 (U = 672).

3.3.2 Cell 4 outflow analysis

Table 9. Cell 4 outflow analysis

	Normal	Recycle	HNF	HFR
Ammonia	H: 14.582 df: 3 p = 0.002	H: 1.415 df: 3 p = 0.702, ns	H: 3.146 df: 3 p = 0.370, ns	H: 1.593 df: 3 p = 0.661, ns
Orthophosphate	H: 1.031 df: 3 p = 0.794, ns	H: 3.849 df: 3 p = 0.278, ns	H: 6.288 df: 3 p = 0.098, ns	H: 5.472 df: 3 p = 0.140, ns
Nitrite	H: 4.386 df: 3 p = 0.223, ns	H: 0.958 df: 3 p = 0.811, ns	H: 4.215 df: 3 p = 0.239, ns	H: 0.919 df: 3 p = 0.821, ns
Nitrate	H: 0.775 df: 3 p = 0.855, ns	H: 0.671 df: 3 p = 0.880, ns	H: 2.354 df: 3 p = 0.502, ns	H: 0.149 df: 3 p = 0.985, ns
TON	H: 0.527 df: 3 p = 0.913, ns	H: 0.415 df: 3 p = 0.031, ns	H: 2.416 df: 3 p = 0.491, ns	H: 0.298 df: 3 p = 0.960, ns
Chloride	H: 1.197 df: 3 p = 0.754, ns	H: 5.447 df: 3 p = 0.142, ns	H: 19.088 df: 3 p = 0.000	H: 1.196 df: 3 p = 0.754, ns
BOD	H: 2.315 df: 3 p = 0.510, ns	H: 6.048 df: 3 p = 0.109, ns	H: 5.526 df: 3 p = 0.137, ns	H: 1.017 df: 3 p = 0.797, ns
PH	H: 11.533 df: 3 p = 0.009	H: 4.332 df: 3 p = 0.228, ns	H: 15.283 df: 3 p = 0.002	H: 15.294 df: 3 p = 0.002
Conductivity	H: 4.437 df: 3 p = 0.218, ns	H: 19.311 df: 3 p = 0.000	H: 7.458 df: 3 p = 0.059, ns	H: 0.292 df: 3 p = 0.962, ns
DO	H: 1.110 df: 3 p = 0.775, ns	H: 3.688 df: 3 p = 0.297, ns	H: 8.197 df: 3 p = 0.042	H: 1.620 df: 3 p = 0.655, ns
Temperature	H: 0.113 df: 3 p = 0.990, ns	H: 0.270 df: 3 p = 0.965, ns	H: 0.437 df: 3 p = 0.933, ns	H: 0.814 df: 3 p = 0.846, ns

Note: Significant tests highlighted.

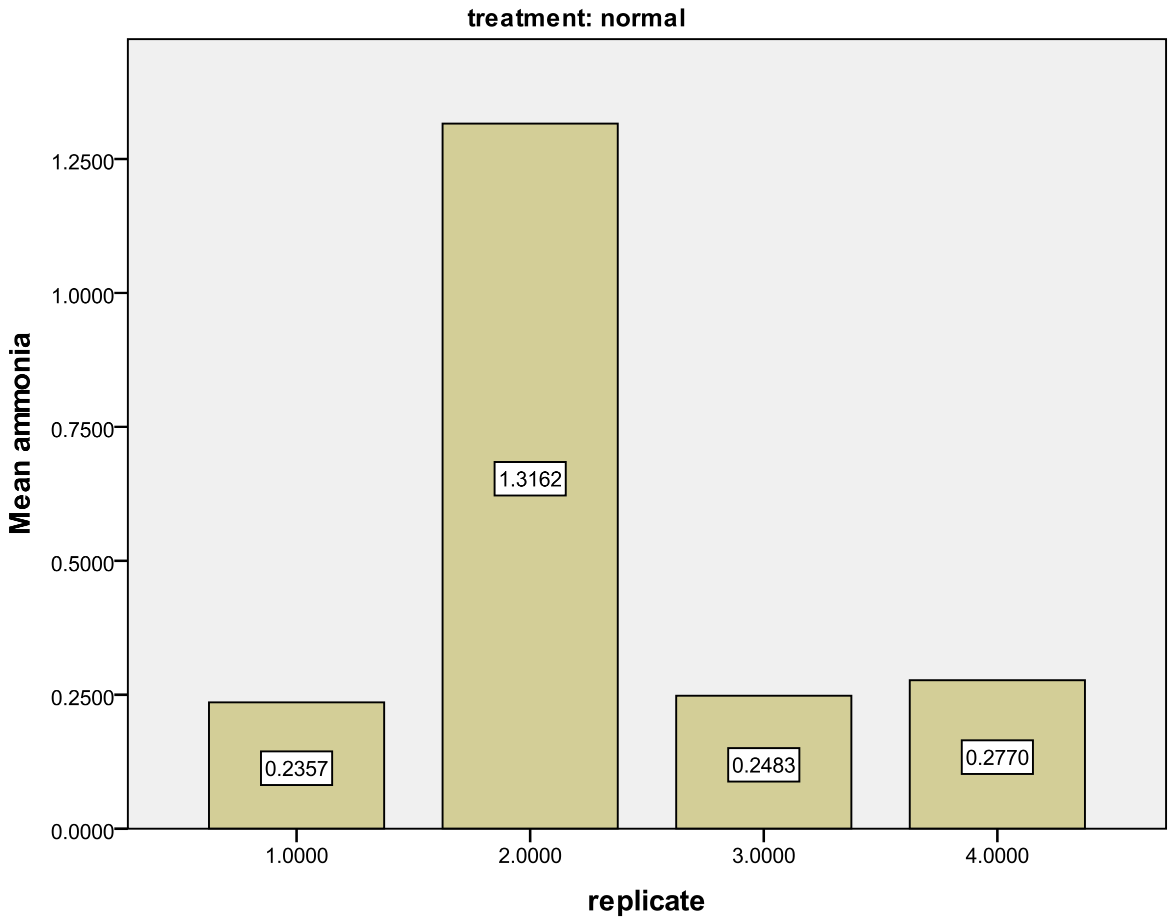

Figure 26. Replicate 2 is significantly different to replicate 3 (U = 573) and replicate 4 (U = 704).

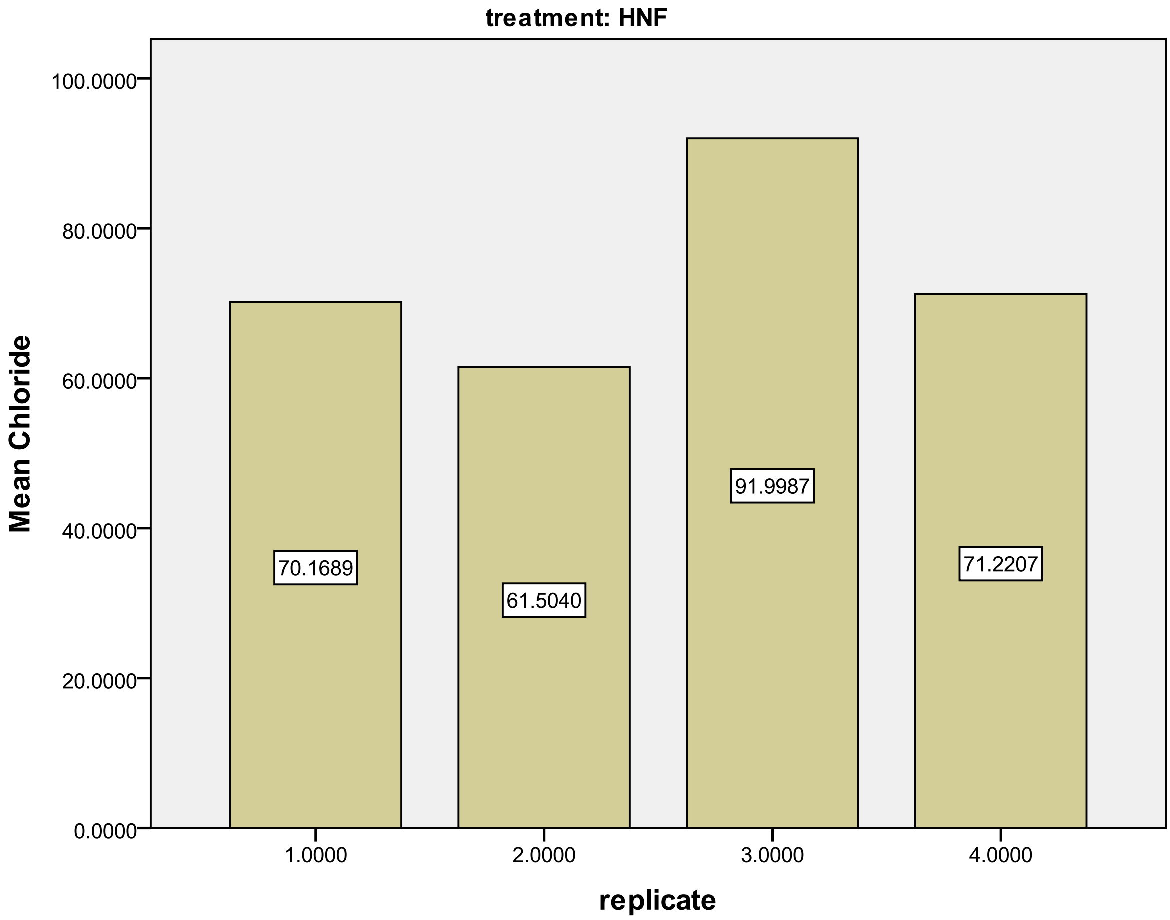

Figure 27. Replicate 3 is significantly different to replicate 1 (U = 611), 2 (U = 510.5) and 4 (U = 613).

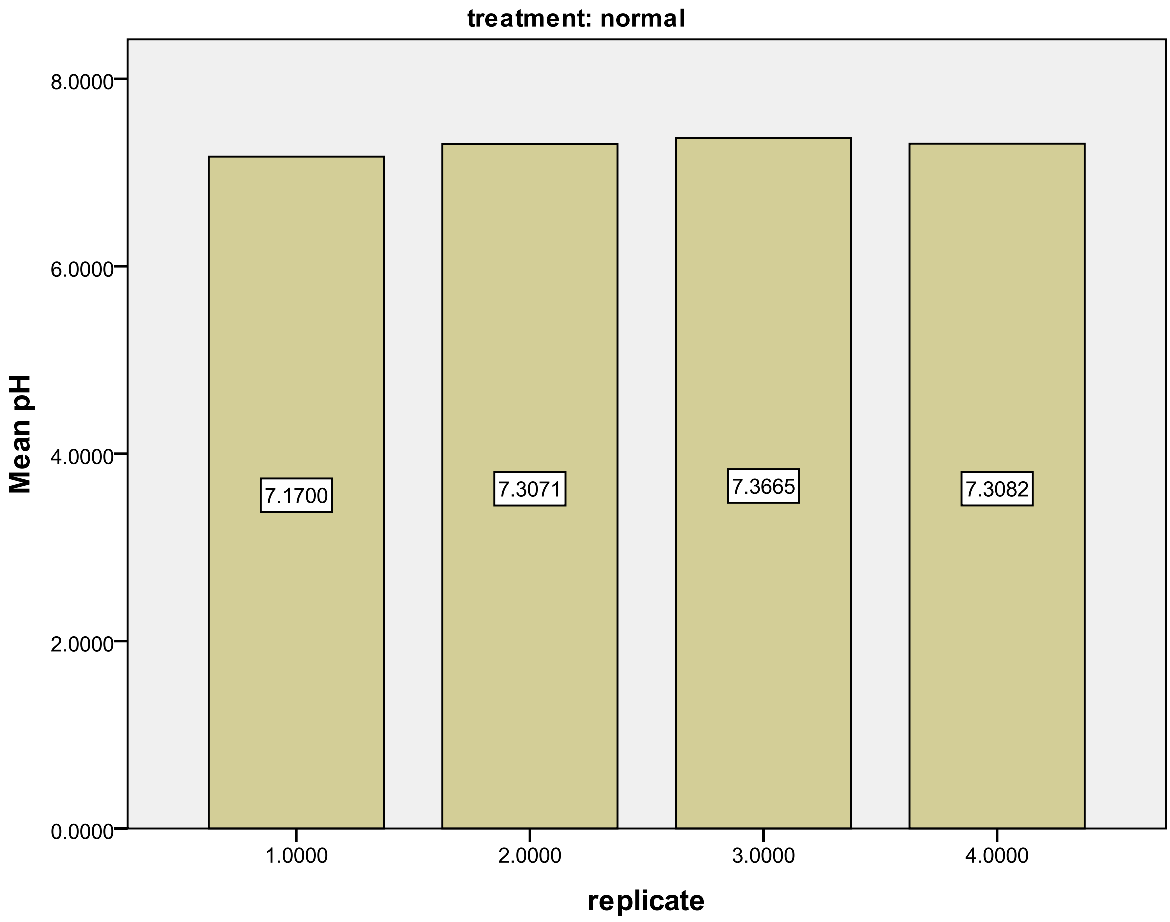

Figure 28. Replicate 1 is significantly different to replicate 3 (U = 57.5) and 4 (U = 73).

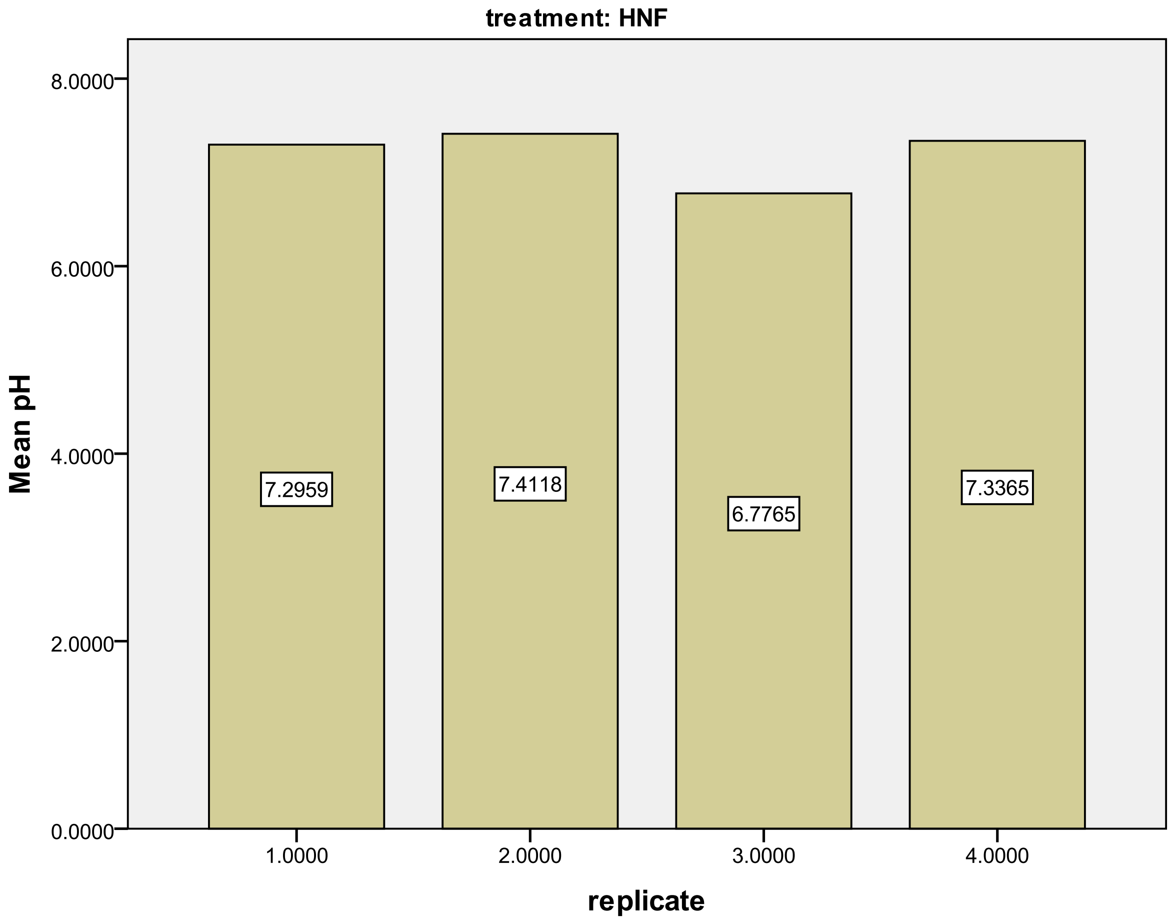

Figure 29. Replicate 1 is significantly different to replicate 2 (U = 72). Replicate 3 is significantly different to replicate 2 (U = 51) and 4 (U = 69.5).

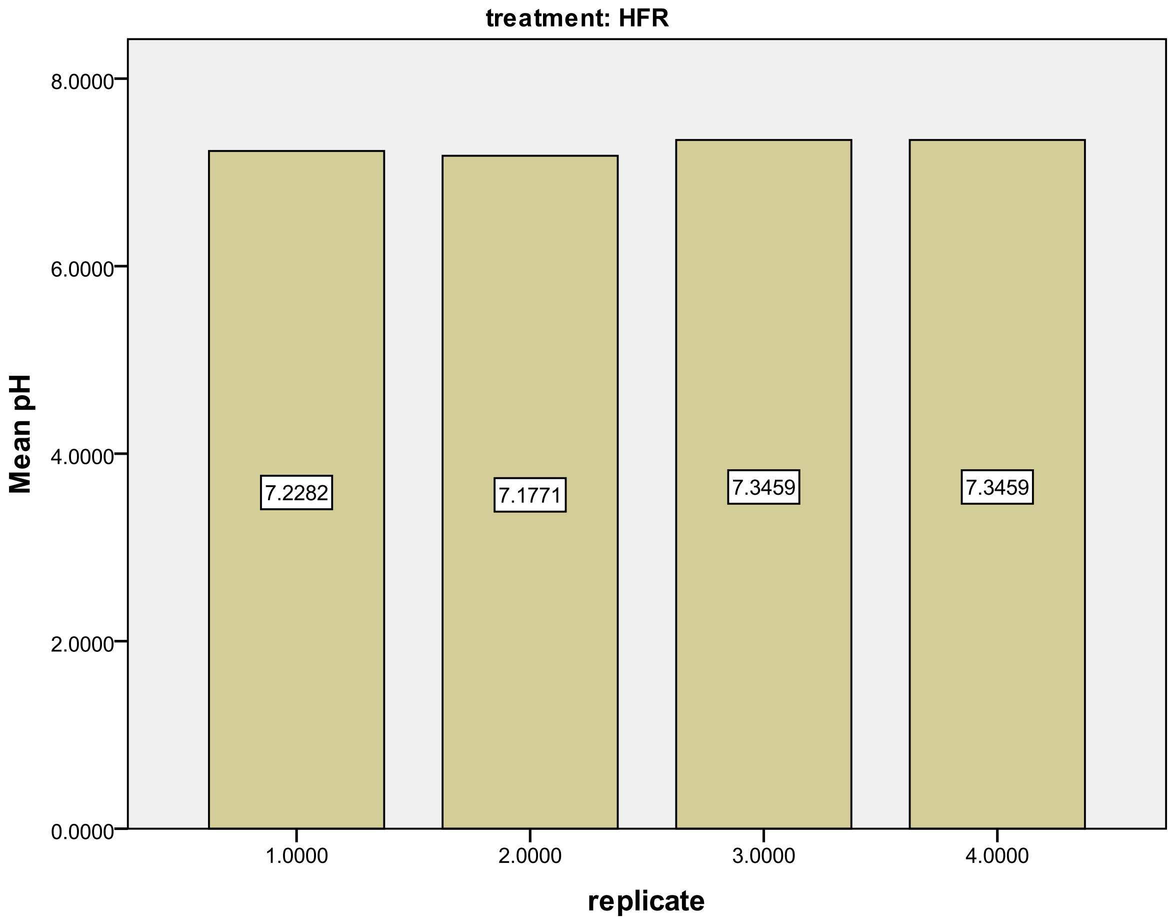

Figure 30. Replicate 2 is significantly different to replicate 3 (U = 56) and replicate 4 (U = 56.5).

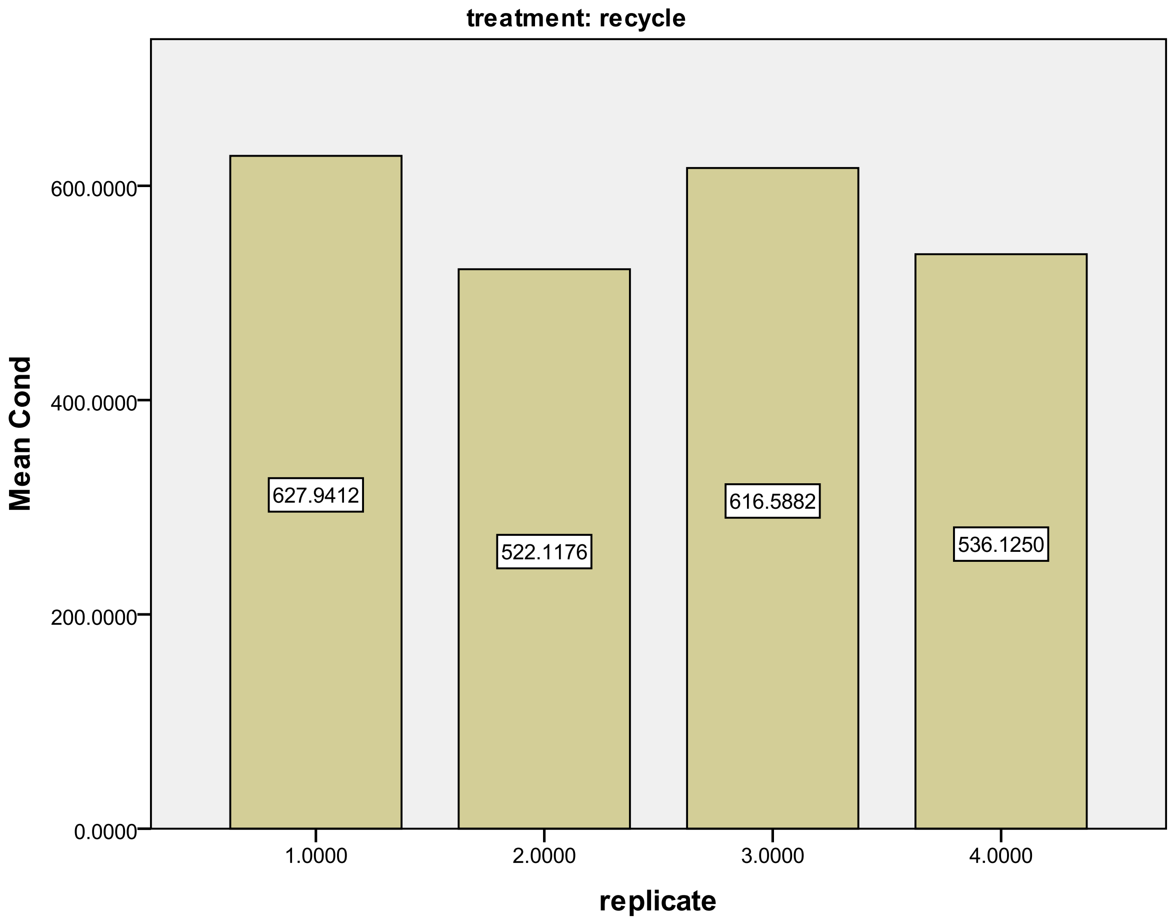

Figure 31. Replicate 2 is significantly different to replicate 1 (U = 27) and 3 (U = 40.5).

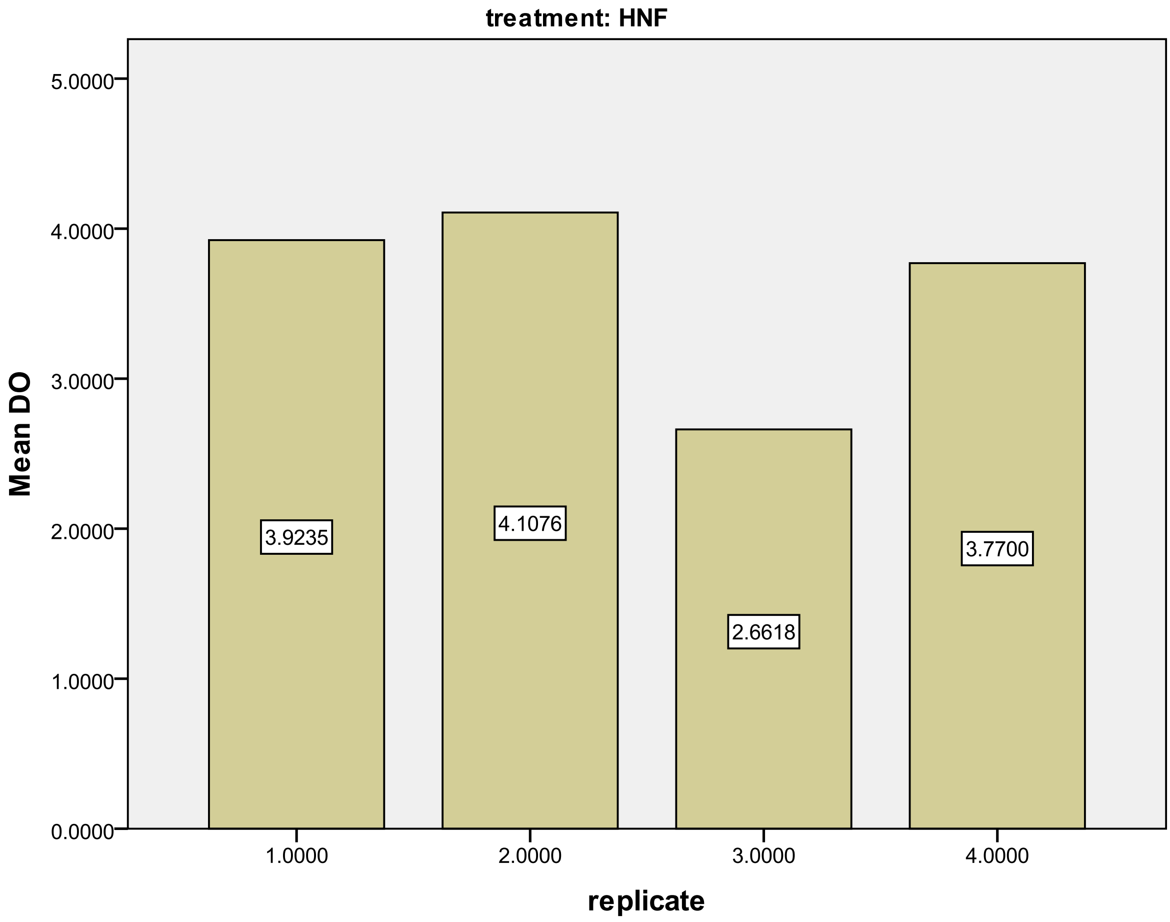

Figure 32. Replicate 2 is significantly different to replicate 3 (U = 67).

3.4 Treatment differences

This section looks for differences between the outflows from differing treatment types. The data from all replicates has been used in this analysis.

Table 10. Differences between outflows from cell 1 and cell 4

	Cell 1 outflow	Cell 4 outflow
Ammonia	H: 216.263 df: 3 p = 0.000	H:80.178 df: 3 p = 0.000
Orthophosphate	H: 82.061 df: 3 p = 0.000	H: 52.876 df: 3 p = 0.000
Nitrite	H: 38.321 df: 3 p = 0.000	H: 92.590 df: 3 p = 0.000
Nitrate	H: 29.184 df: 3 p = 0.000	H: 48.826 df: 3 p = 0.000
TON	H: 32.920 df: 3 p = 0.000	H: 57.245 df: 3 p = 0.000
Chloride	H: 54.345 df: 3 p = 0.000	H: 108.145 df: 3 p = 0.000
BOD	H: 24.938 df: 3 p = 0.000	H: 14.001 df: 3 p = 0.003
PH	H: 43.315 df: 3 p = 0.000	H: 4.870 df: 3 p = 0.002
Conductivity	H: 02.031 df: 3 p = 0.000	H: 133.319 df: 3 p = 0.000
DO	H: 31.568 df: 3 p = 0.000	H: 11.932 df: 3 p = 0.008
Temperature	H: 4.032 df: 3 p = 0.258, ns	H: 3.351 df: 3 p = 0.341, ns

Note: Significant tests highlighted.

3.4.1 Cell 1 outflow treatment differences

Table 11. Comparative table between cell 1 outflow from all treatment methods

Comparisons:	**Normal Recycle**	**Normal HNF**	**Normal HFR**	**Recycle HNF**	**Recycle HFR**	**HNF HFR**
Ammonia	ns	5669	7336.5	4525	5765	11344
Orthophosphate	ns	8724	11101	9022	11389	13381.5
Nitrite	12692	ns	ns	10600.5	11332	ns
Nitrate	ns	13109	ns	11186	12750.5	ns
TON	ns	13381	ns	10975	12068.5	ns
Chloride	11663	12706	ns	ns	9577	10759
BOD	ns	ns	859.5	739	590.5	ns
PH	ns	1079.5	ns	1104	ns	1092
Conductivity	ns	381.5	1069.5	520.5	1246.5	942
DO	ns	1657	1212	ns	1242	ns
Temperature	ns	ns	ns	ns	ns	ns

Note: table contains test statistic values (U) for significant differences. A Bonferroni correction was applied and so all effects are reported at a 0.008 level of significance.

3.4.2 Cell 4 outflow treatment differences:

Table 12. Comparative table between cell 4 outflow from all treatment methods

Comparisons:	Normal Recycle	Normal HNL	Normal HFR	Recycle HNL	Recycle HFR	HNL HFR
Ammonia	13318.5	12550.5	11296.5	9227	8323.5	ns
Orthophosphate	11785.5	9604	10017.5	ns	ns	ns
Nitrite	ns	11906.5	9465.5	10078	7738	ns
Nitrate	ns	13370	10741	12698.5	10049	13371
TON	ns	12998.5	10422.5	12146	9546.5	13361.5
Chloride	ns	7182	11289	8182.5	12495	10519
BOD	820.5	ns	ns	856.5	ns	ns
PH	1439	ns	ns	ns	1355.5	ns
Conductivity	1421.5	538	122.5	798	314	ns
DO	ns	ns	ns	ns	1472	ns
Temperature	ns	ns	ns	ns	ns	ns

Note: contains test statistic values (U) for significant differences. A Bonferroni correction was applied and so all effects are reported at a 0.008 level of significance.

Table 13. Cell 4 outflow

	Normal	**Recycle**	**HNF**	**HFR**
Ammonia	H: 14.582 df: 3 p = 0.002	H: 1.415 df: 3 p = 0.702, ns	H: 3.146 df: 3 p = 0.370, ns	H: 1.593 df: 3 p = 0.661, ns
Orthophosphate	H: 1.031 df: 3 p = 0.794, ns	H: 3.849 df: 3 p = 0.278, ns	H: 6.288 df: 3 p = 0.098, ns	H: 5.472 df: 3 p = 0.140, ns
Nitrite	H: 4.386 df: 3 p = 0.223, ns	H: 0.958 df: 3 p = 0.811, ns	H: 4.215 df: 3 p = 0.239, ns	H: 0.919 df: 3 p = 0.821, ns
Nitrate	H: 0.775 df: 3 p = 0.855, ns	H: 0.671 df: 3 p = 0.880, ns	H: 2.354 df: 3 p = 0.502, ns	H: 0.149 df: 3 p = 0.985, ns
TON	H: 0.527 df: 3 p = 0.913, ns	H: 0.415 df: 3 p = 0.031, ns	H: 2.416 df: 3 p = 0.491, ns	H: 0.298 df: 3 p = 0.960, ns
Chloride	H: 1.197 df: 3 p = 0.754, ns	H: 5.447 df: 3 p = 0.142, ns	H: 19.088 df: 3 p = 0.000	H: 1.196 df: 3 p = 0.754, ns
BOD	H: 2.315 df: 3 p = 0.510, ns	H: 6.048 df: 3 p = 0.109, ns	H: 5.526 df: 3 p = 0.137, ns	H: 1.017 df: 3 p = 0.797, ns
PH	H: 11.533 df: 3 p = 0.009	H: 4.332 df: 3 p = 0.228, ns	H: 15.283 df: 3 p = 0.002	H: 15.294 df: 3 p = 0.002
Conductivity	H: 4.437 df: 3 p = 0.218, ns	H: 19.311 df: 3 p = 0.000	H: 7.458 df: 3 p = 0.059, ns	H: 0.292 df: 3 p = 0.962, ns
DO	H: 1.110 df: 3 p = 0.775, ns	H: 3.688 df: 3 p = 0.297, ns	H: 8.197 df: 3 p = 0.042	H: 1.620 df: 3 p = 0.655, ns
Temperature	H: 0.113 df: 3 p = 0.990, ns	H: 0.270 df: 3 p = 0.965, ns	H: 0.437 df: 3 p = 0.933, ns	H: 0.814 df: 3 p = 0.846, ns

Note: Significant tests highlighted.

4. Discussion

The use of constructed wetlands for the treatment of various wastewaters is not a new approach, but the refinement of their use is on-going. The premise of this study was to take an approach, that of the Integrated Constructed Wetlands (Harrington & McInnees, 2009; Scholz *et al*., 2007, Harrington & Ryder, 2002), and attempt to adapt it for the treatment of swine wastewaters.

Taking a linearly-scaled sandbox approach, heavily influenced by the writings of Bormann regarding the Hubbard-Brook Sandbox experiment (Bormann *et al*., 1987), was deemed the most logical approach to the design and operational remit for the project, because it allowed well-defined parameters to be simply scaled down to a size that was practically and financially manageable, as well as being able to maintain a reliable sampling and analysis procedure.

What must be highlighted in regard to the initial data from the systems is that the meso-scale ICW systems were made operational late in the year (December 2008). The growing season had ceased and the plants, being semi-deciduous, had begun dying back for the winter and their potential to adapt to the influent loadings and concentrations was directly affected. The initial removal rates, whilst promising, were short-lived and the notable drop in performance was not only due to over-loading of nutrient concentrations and volumes, but also in that both the microbial and vegetative components of the systems were not mature enough to cope with the higher influent material. The lowering of the hydraulic loading into each system by 33% initially was an attempt to overcome this, and the subsequent additional reduction of 33% reduced the loadings to a level that was sustainable throughout the year. While the influent concentrations were remarkably low compared to 'real-world' swine wastewater levels (diluted 1:32), they were deemed appropriate for study purposes as a study of this nature and scale of replication has not been previously operated in Ireland. With little literature available on the use of constructed wetlands in Ireland treating swine wastewaters, the influent levels were kept to a safe concentration; similar to what is commonly seen in full-scale ICW systems in use in Ireland. At the time of design and construction, there were no other similar systems anywhere else in Europe.

The initial increase in nutrient removal performance after the first hydraulic loading adjustment was notable and, after several weeks, removal rates began to reach levels that are comparable to those that are sought in full-scale ICW systems in regard to discharge licensing, but they were still deemed to be relatively weak. The second decrease in hydraulic loading resulted in effluent concentrations that met or exceeded drinking water standards in Ireland for the majority of the sampling period. The loading may have been run at 66% of the original rates, but after having made two changes to the systems already, it was decided between the author and supervisors that the rates of $37m^3$/ha/day and $74m^3$/ha/day were acceptable and thus they were maintained for the remainder of the study period.

The use of the data from 10^{th} of June 2009 to 10^{th} of June 2010 provided a comprehensive 12 month sampling period and a distinct breakdown of the seasonal performance of the systems following maturation and adaptation to the influent liquid. *Gylceria maxima* is a common and widespread wetland plant, but rarely is it examined exclusively in constructed wetland systems, where others such as *Phragmites australis*, *Typha* sp. and *Carex riparia* are the more commonly found. The use of *Glyceria maxima* was primarily based upon personal field observations and practical experience and literature (Tanner, 1996) in wetland systems where

Glyceria had demonstrated a very clear ability to deal with a varying influent concentration and hydraulic loading.

4.1 Initial removal of nutrients

Initial removal of nutrients from the influent liquid demonstrated positive results with effluent quality reaching levels below the Irish Drinking Water Standard of 0.5mg/l. However this was short-lived and the removal rates reduced after four weeks of operation. Between the 10th of January and the 10th of March, the effluent ammonia-N concentrations continued to increase. The lowest recorded removal rates were on 25th February 2009, where the average ammonia-N removal rates were 61.79%, 67.22%, 41.23% and 44.11% for each treatment respectively. This was most likely due to a combination of two factors; hydraulic loading being too high and start-up of the systems too late in the year. There was two months of lead-in time after the construction of the meso-scale systems where each system was filled with tap- and rain-water. This time was used to allow the macrophytes to establish and root into the substrate material.

Glyceria maxima is a deciduous plant, in that it dies back every winter and its transpiration, growth and uptake rates are substantially reduced. Within each system, the majority of the plants had died back by the end of December 2008 to a short stem with most of the extended foliage dead and accumulating as detritus. This decreased period of senescence over the winter months can have an impact on nutrient removal and uptake however, studies examining the nutrient uptake of plants as well as the monitoring of large-scale systems have recorded that nutrient removal efficiencies in constructed wetlands during winter months is negligible (Vymazal, 2011; Scholz *et al*, 2010), though some Asian studies have shown that winter removal rates to have a pronounced decrease (Hu *et al*, 2010). However, in the highly monitored ICW systems in the Anne Valley project in Co. Waterford, Ireland, their removal rates remain relatively high throughout the year (Scholz *et al*, 2007; Babatunde *et al*, 2008; Harrington and McInnes, 2009).

The decreased nutrient uptake by the macrophytes was compounded by the initial hydraulic loading. The initial loadings were set at approximately 112m^3/ha/d and 200m^3/ha/d. Higher influent loadings coupled with winter rainfall of 54.9mm in the first three weeks of December 2008 (Met Eireann) decreased hydraulic retention periods in the systems and each treatment had a substantial outflow volume (>60 L/week).

Table 14. Rainfall amounts during highflow period of sampling.

Date	**Rainfall (mm)**	**% monthly average**
December 2008	54.9	50
January 2009	212.6	182
February 2009	14.4	17
March 2009	50.5	64

4.2 Ammonia reductions

The ammonia removal rates for the meso-scale systems were similar to other full-scale ICW systems that have been in operation in the South-east of Ireland for the past decade (Harrington and Ryder, 2005; Scholz *et al*, 2007; Babatunde *et al*, 2008; Harrington and McInnes, 2009). Throughout the 18 months of operation, removal rates from the meso-scale systems remained relatively stable regardless of seasonal changes. This stability was also recorded among the replicates . Ammonia concentrations in the final effluent of each treatment remained low between June 2009 and June 2010. Normal treatment showed an average concentration of 0.53 mg ammonia/L with a standard deviation of ±0.6. Recycling, HNL and HFR treatments showed similar ammonia concentration levels during this time period of 0.17mg NH_3/L, 0.69 mg NH_3/L and 1.94 mg NH_3/L respectively.

Over the 18 months that the meso-scale systems were in full operation, the recycling treatments showed the lowest ammonia concentrations in the final effluent (0.005 mg/l). The improvement in effluent quality when recycling has been implemented has been seen in previous studies (Kantawanichkul *et al*., 2009). The increase in nitrified water promotes greater denitrification and the recirculation of the effluent back into the system increases the retention time of the system. This has also been demonstrated in other studies where land availability is limited or the area allowed for the treatment wetland is restricted (Kantawanichkul *et al*, 2009; He *et al*, 2006). The HFR system showed significantly higher corresponding concentrations for Ammonia-nitrogen. Summer months showed very low volumes of water being collected in the overflow containers from all four operations, with high outflow a direct result of heavy rainfall, which provided additional dilution of the effluent liquid.

4.3 MRP reductions

Molybdate reactive phosphorus (MRP) removal was also very high in all four treatment systems averaging 97.7%, 96.0%, 94.6% and 89.1%, for Normal, Recycling, High Nutrient Loading and High Flow Rate respectively. These removal rates are in line with other full-scale ICW systems that are currently in operation in Ireland treating dairy, cattle, municipal and food-production wastewaters (McInnes *et al*., 2012; Babatunde *et al*., 2008; Scholz *et al*., 2007).

The MRP levels in the influent were substantially reduced after separation following anaerobic digestion as most had been removed in the solid fraction. The main mechanisms for phosphorus removal are adsorption and sedimentation (Kadlec, 1999).

MRP concentrations in the final effluent maintained estimated discharge limits for Ireland (<1mg P/l) throughout the operational timeframe, including the higher hydraulic loading phase. Discharge licensing in Ireland is granted on a site by site basis and is dependent upon the assimilative capacity of the receiving water body. Despite the systems only having a washed aggregate substrate, the removal efficiency remained on a similar level with fully operational and established ICW. These rates were maintained during the winter periods, showing year-long effectiveness. Prolonged usage with a limited substrate could potentially yield lower removal rates (Dunne *et al*., 2002), however, this has not been seen in ICW systems still in operation in Ireland for almost a decade. The influent MRP concentrations over the 18-month period were an average 3.13mg P/l and 3.25mg P/l for the 12-month period. The effluent concentrations in Normal and Recycling frequently went below that of the limits of detection (<0.01mg P/l), however such low concentrations were rarely seen in HNL and HFR systems.

4.4 Mass nitrogen removal

When the potential practical nature of ICWs, is considered, the mass nitrogen removal is a relevant parameter to examine given the integration of water quality, landscape fit and biodiversity (Harrington and McInnes, 2009). In Ireland, the

Nitrates Directive limits the amount of organic-N that can be applied to spreadlands as 170KgN/ha/yr. In many instances where the farmer has limited spreadlands, does not have the option of providing material to neighbouring farmers or does not have sufficient storage capacity, the use of constructed wetlands may provide a sufficient method of balancing the nitrogen budget for the farm Nolan *et al*., 2011. In the case of piggeries where spreadlands are highly limited, the ability of constructed wetlands to remove large amounts of nitrogen provides an opportunity for the farmers to balance their nitrogen budget more effectively.

The meso-scale systems were examined for their mass nitrogen removal efficiencies over the 12-month sampling period. Extrapolation of the results demonstrates that the systems had the potential to remove 127kg nitrogen/ha/yr in the Normal, Recycling and HNL and up to 230kg nitrogen/ha/yr in the HFR system. Reinhardt (2006) documented an average nitrogen removal of 45kg nitrogen/ha/yr. In periods where the stored influent material was not sampled, the storage tank data were extrapolated to get a best fit and a linear extrapolation was used to determine the storage tank ammonia-N concentration. This was then used in the calculations to obtain the mass removal capacity of the treatments.

With the Nitrates Directive placing a rigid limit of 170Kg of organic-N/ha/yr for spreadlands, the potential ability for an ICW to remove up to 230kg/ha/yr of Nitrogen yields opportunities for farmers. For instance, a 500 sow integrated piggery unit would require almost 41 hectares of spreadlands so as not to exceed the 170kg limit (S.I. No. 610). A potential balance could be made between treating piggery wastewaters and the use of them as fertilizers on available grass or crop lands, allowing for a more manageable nitrogen management system for a piggery (Nolan *et al*., 2012). In 2013 the Nitrates Directive will be expanded and landspreading quotas will be taking Phosphorus concentrations into account, this may yield additional problems for farmers/landowners that are already heavily loaded with organic and inorganic P.

4.5 Pathogen reductions

During the operation of this project, additional research was run in parallel by Gemma McCarthy from Waterford Institute of Technology, as part of the Stimulus Research Funding Project, as described in section 2.9. This research was run in conjunction with the fieldwork that was undertaken on the meso-scale wetlands and sampling was performed at the same time, so as to have accurate corresponding results from the meso-scale wetlands. (McCarthy *et al*, 2011).

Both the fieldwork on the full-scale systems and the meso-scale systems had shown over the sampling period that they were capable of significantly reducing indicator micro-organisms to the levels of detection. The full-scale systems were only sampled during either March and May, 2010 or May and June, 2010. These two sampling periods were the only sampling done for the large-scale system, whereas the meso-scale systems were sampled monthly for the period between April 2009 and May 2010 (13 months). Salmonella was not detected in the influent material at any stage during the 13 months of sampling. Additionally, insufficient levels of E. coli and Enterococcus in either the influent or any effluent samples to allow for reliable conclusions to be made on the recorded data. However, coliform were present in higher counts in the influent samples with the samples from the individual cells through the sampled systems showing mean counts of coliforms to be generally unaffected by any treatment parameter. Only in Normal treatment was there a recorded slight decrease in Coliform counts (5.8%). Some linear reductions were recorded through the cells in regard to Coliforms between cell 1 and cell 2 in Normal operation systems. However there was no further reduction from the effluent of cell 2 and cell 3. Aerobic spore-forming bacteria were recorded as being reduced in a linear manner, however in comparison to Coliforms, the reduction was not recorded until cell 3 rather than cell 2. Seasonal variation yielded no significant changes to the removal rates across any of the treatment methods, despite some recorded difference.

Overall, the microbial analysis with regard to the meso-scale ICW systems was for the most part inconclusive due to counts being too low to use effectively, such as *E. coli* and *Enterococcus* and the lack of any recorded Salmonella in the influent or any other sample. The recording of yeasts, spore-forming bacteria and coliforms provides a superficial insight in regard to the ability of these meso-scale systems in being able to treat them, but the results overall remain inconclusive. Compared to the full-scale, mature ICW systems that were sampled in 2010 (McCarthy *et al*., 2011), it is difficult to draw direct comparisons between the two different scales, though there are some similarities with currently unpublished data.

The separation of the solid and liquid fractions in conjunction with the small scale of the meso-scale ICW may have an effect on the tests that were applied to the meso-scale samples. Despite there being a recorded drop in spore-forming bacteria, yeasts and coliform in the meso-scale ICW systems, only coliform are considered to be enteric indicators. The study concludes that additional sampling would be required to form a more definitive result and conclusion with the possibility of requiring antibiotic-marked strains which would facilitate improved pathogen tracking.

4.5 Stability of the systems

Considering an exceptionally harsh winter and spring in 2010, the meso-scale systems encountered very few problems with regard to maintenance and operation or their treatment capacity. Indeed the effectiveness of CW has been shown to work in cold climates, such as Finland, Estonia, Canada and Sweden (Poldvere *et al*, 2009; Maltais-Landry *et al*, 2009; Basviken *et al*, 2009; Puustinen & Jormola, 2005). Any electro-mechanical components that did fail were easily replaced and with minimal, if any, downtime. The high level of replication of the treatments gives great statistical confidence in the data collected. Any significant variability between expected and recorded results was easily identified and the cause could be identified from the replicate itself by comparison with other replicates. Generally, two replicates are common in most CW studies (Bastviken *et al*, 2009; Hunt *et al*, 2002) however some have used very high replicate counts when examining more biological parameters, such as mosquito populations (Poach *et al*, 2004). The sizing of these systems when viewed in terms of the materials used is advantageous as it allows for high replication whilst minimizing the required land area. At the same time, it allows for the sizing of these systems to be relatively large in comparison to bench-scale tests. Throughout the operation of the meso-scale systems, very few problems were encountered and those that did occur, such as blockages of the siphon traps and pump failure were easily corrected with little or no downtime to the systems themselves. The high replication of each treatment allowed for great confidence in the recorded results throughout the sampling period. While this did come at a relatively small cost with respect to required land area, it ensured reliable results and the ability to examine the systems themselves.

4.6 Simple mechanics and reliability

The use of the simple construction materials and equipment was paramount in the design, second to that of mimicking the key aspects of the aspect ratios of the

ICW concept as laid out in the literature, primarily Scholz *et al*, 2007, DoEHLG, 2010 and Carty *et al.*, 2008.

Using simple materials allowed for the system to be repaired very quickly with minimal complications, but also was a way of examining the feasibility of using such materials and equipment that had been until then, untested in such circumstances. With small repairs needing to be made from time to time throughout the complete operation of the meso-scale system, the relative ease of being able to acquire parts needed in any local hardware store made it all a much more manageable design. Winter periods of severe cold during 2009 and 2010 resulted in the pumps in both tanks breaking down due to freezing of the stored liquid which caused the motors to burn out as they were self-priming. During both winters, at least one pump ceased to be operational and all pumps were replaced as a precautionary measure. No other components in the entire meso-scale system needed repairs or replacement at any stage during the study period. The small clearance of minor blockages in the siphon traps being cleared after periods of downtime during the freezing temperatures was the only exception.

4.7 Comparative performance

The recorded data from the 18 month sampling period was broken essentially into three periods; High flow, low flow and 12-month. High flow relates to the initial pumping of material into the system at the higher hydraulic loading rates of 117m^3/ha/d and 147m^3/ha/d from December 2008 to March 2009. Low flow was the period beginning March 2009 until completion where the hydraulic loading rates were at 37m^3/ha/d and 74m^3/ha/d. The 12-month period relates to the 6th of June 2009 until 6th of June 2010.

Average ammonia-nitrogen removal rates from each system during the high flow period were 83.6%, 86.4%, 57% and 76.2% for Normal, Recycling, HNL and HFR respectively, between 10th December 2008 and 8th April 2009. These initial high removal rates included a lag period from when the influent material was first introduced into the systems. These rates were considered poor in comparison to full-scale ICW (Babatunde *et al*., 2008; Scholz *et al*., 2007) and the influent hydraulic loading was reduced eventually to the 12-month rate of $37m^3$/ha/d and $74m^3$/ha/d respectively. The resulting increase in ammonia-nitrogen removal performance gave removal averages of 99.5%, 99.9%, 99.7% and 98.1% across the four treatment systems (Harrington *et al*, 2011).

While the change in influent loading improved the effluent quality in each system, there were fewer parameters that showed a significantly different result than the effluent from cell 1. Ammonia-nitrogen levels from Normal treatment cell 4 were seen to be significantly different to the other systems ($p<0.05$). Chloride was also shown to be significantly different ($p<0.05$) in HNL cell 4 in comparison to the other system. Several systems showed substantially different values under the parameters of pH, conductivity and dissolved oxygen (Table 14).

The lack of significantly different values obtained from the final cells of each treatment indicates the ability of each system to adequately treat nitrogen- and phosphorus-species. The higher value given for ammonia-nitrogen in Normal treatment is due to increased levels of ammonia-nitrogen in replicate 2 for a period of 15 weeks between 5th November 2009 and 8th April 2010.

Similarly the increased value for chloride given for HNL are the result of the higher concentration of influent material into those systems and this carrying through the cells, most notably in replicate 3 between 22nd July 2009 and 13th August 2009 where the effluent chloride levels were higher than the cell 1 levels and ranged between 124.97mg/l and 179.96 mg/l. Indeed, on the 3rd of June 2010, effluent levels exceeded 200 mg/l in replicates 3 and 4 and reached 197 mg/l.

The lack of distinctly comparative nutrient removal rates between the systems highlights the effectiveness of the systems, regardless of operational parameters when dealing with low concentration material and low loading. The presence of three significant parameters in a single treatment, HNL, hints at there being potential problems with higher rates under those parameters. Substantial increases of nutrient concentrations and hydraulic loading could have a detrimental effect on both the wetland and its treatment capacity. However, the HNL systems also yielded better nutrient removal rates than HFR, which only had significantly different pH values assigned and even that was only in relation to Recycling.

No single treatment system was recorded in having significantly different removal rates or concentration levels, when viewed against discharge licensing in Ireland, as all treatments produced effluent with very low concentrations of all parameters. Recycling was the only treatment to consistently produced effluent with a value below 0.5 mg/l NH_3-N. However, when the percentage removal rates are examined, all four treatments achieved greater than 98% removal for ammonia-N and greater than 89% for MRP. If volume is taken into account, Recycling produced the least amount of effluent volume, with three months of 2009 having zero discharge at all, though some other replicates had this sporadically as well. The increased hydraulic retention time as a result of the recycling mechanic would potentially allow for increased nutrient reductions.

HFR showed a significant difference in terms of its ammonia concentrations, which is likely due to the shorter retention time in those systems. Orthophsophate, chloride and BOD_5 also showed significant differences between various treatments (Table 9). The range of significantly different values across the systems was not repeated in the data from cell 4. Table 9 shows that there was less than half the number of significant differences between treatments than were in Table 9 (cell 1 outflows). The most reoccurring difference is in regard to pH values and conductivity. Overall, there is very little in the way of statistically significant differences between the systems but also in a more practical and 'real-world' manner. The effluent concentrations for ammonia-N regularly reached if not surpassed drinking water standards for Ireland (>0.05mg/l). When viewed objectively against the empirical data that is given in tables 7 & 8 and shown fully in appendix 5, the key nutrient parameters (ammonia & MRP) generally all have exceptional reduction rates. These removal rates were similar across the replicates within each treatment as well. With regard to the nutrient rates from cell 4, only ammonia in the Normal treatment systems had a single replicate that was significantly different to the other replicates within that treatment. This replicate (replicate 2) did have higher ammonia-nitrogen concentrations than the other replicates for an extended period of time during the field work. With the exception of this singular replicate, there are no significant differences between any replicates with regard to ammonia, MRP, nitrate and nitrite. This level of similarity gives absolute confidence in terms of comparative analysis of the treatments and their respective replicates performance. Statistical analysis across all four treatment methods (Table 13) shows that there are some significant differences between the treatments, but not in an encompassing manner. There are some parameters which are noted as being significantly different to others, but when viewed and compared against the collected data (Appendix 5) this difference may not actually be positive. Between ammonia, orthophosphate, nitrite, nitrate and TON, only Normal with regard to ammonia-N reduction was shown to be significantly different to the other treatments. Whilst no single treatment parameter is shown to be superior to any of the other treatments, section 4.9 addresses the shortcoming with regard to this in term on influent concentrations. Should the concentrations have been higher, with the stresses put on the systems being potentially much more detrimental, there is indeed the possibility that one system may have come through as a distinctly superior treatment method.

The HNL and HFR systems had visually healthier macrophytes, in terms of density, colour and height, due to the higher influent concentrations that allowed for more nutrients to reach the macrophytes in later cells. Comparatively, the Recycling and Normal systems had visually poorer macrophyte growth in cells 2, 3 and 4 with much greater quantities of algae and other flora becoming present throughout the study period. Recycling and Normal treatments had lower effluent volumes in comparison to HNL and HFR and when there was a discharge present, the nutrient concentrations were slightly less than what HNL and HFR would yield, most likely as a result of the increased retention times and lower loadings (both nutrient and hydraulic).

4.8 Costs and Land-use

One of the largest concerns with ICW designs to landowners is the relatively large land area required to provide adequate treatment of wastewaters (Everard *et al*., 2012; Harrington & McInnes, 2010; Scholz *et al*, 2007). In relation to piggeries, this would be no less of an issue. In fact it is more likely to be an issue of greater concern due to their limited available spreadlands (Nolan *et al*., 2012). The meso-scale systems were linearly scaled down based upon systems that were already in operation at the beginning of this research project and on available literature, such as

Scholz *et al* 2007. This was the most practical manner in which to approach the issue of sizing using available materials. Harrington and McInnes (2009) stated that the sizing of an ICW system for the treatment of livestock wastewater is primarily determined by the volumetric flow through each section of the wetland itself. Scholz *et al* (2007) discussed the value of using a 4-cell approach, similar to the scaled-down approach applied in this project.

The relatively large area can cause significant issue with landowners and farmers, as many would potentially see this as a loss of available lands for the effective spreading of wastewaters. A Teagasc Piggery Industry Report (Teagasc, 2008) stated that indeed many piggeries would transport their excess wastewaters to nearby farmers for spreading on croplands, but even this has been curtailed as a result of the Nitrates and Water Framework Directives. Harrington *et al* (2011) stated, though based on unpublished documentation, that a surface area of 120-140m^2 per sow would be required in an integrated sow unit where all wastewaters, including washwaters, would enter the ICW system. For an integrated sow unit of 500, this would equate to 6 hectares.

A study was done by Teagasc, which involved research data from this project which examined the cost of treatment of pig manure (Teagasc, 2008), through various methods of treatment which included:

- Anaerobic co-digestion
- Composting
- Liquid and solid separation
- Treatment through an ICW

These were all experimental treatments, except for that of separation as this was performed as part of the base identification of the liquid and solid fractions of the pig manure. The material that was being examined through the ICW is that which is described in this manuscript and was diluted and operated at the levels as described in the materials and methods section. Nolan *et al*, (2012) derived a result that was fixed on the liquid separation of the manure and maintained an influent ammonia-nitrogen concentration level of 200mg/l. As a direct result of the low influent concentrations, the ICW sizing was calculated to be a total of 7.2 hectares. Whilst this is 1.2 hectare greater than the 6 hectares based on the Harrington *et al*. (2011) values, it is still a substantial area and one that a small-scale piggery would not be willing, generally, to 'give up' for the implementation of an ICW. The costings paper (Nolan *et al*., 2012 used the liquid fraction at a concentration of 200 mg/l NH_4 with a hydraulic loading rate of 74 m^3/ha/d. Nolan *et al* (2012) that increasing the influent concentration to 400mg/l would half the land area required for effective treatment. Costs associated with the ICW approach (Nolan *et al*., 2012) include the running of a separator which could increase the cost to the farmer up to €182,469 per annum. This figure is indeed representative of the implementation of a separator as part of the treatment process as a whole, but if it were removed, the annual cost to the farmer/landowner decreases substantially to €48,342. These figures, both land area and costs, can be considered as a theoretical maximum for the use of an ICW for the treatment of piggery wastewaters based on published literature. Additional analysis of the data from this research project would lead to the conclusion that both the land area required and the costs of both constructing and operating a full-scale ICW would be considerably less than previously reported.

When the a nitrogen value for pig manure of 4.2kg/m^3 (S.I No. 610) is used in conjunction with the ammonia-nitrogen percentage of 68% of total Nitrogen being ammonia-nitrogen (Carney *et al.*, 2011), the following equations come to a substantially smaller land-area requirement for a 500 sow integrated unit.

$$4.3 \text{ kg N/m}^3 \text{ (S.I. No. 610)}$$

$$NH_4 = 68\% \text{ TN} = 2.856 \text{ kg/m}^3$$

$$2.856 \text{ kg nitrogen/m}^3 = 2{,}856 \text{ mg/l}$$

If the influent concentration is increased to 600mg NH_4/l a dilution rate of 1:4.76 would be needed.

$$2{,}856 / 4.76 = 600 \text{ mg/l}$$

$$10{,}500 \text{ m}^3\text{/annum raw manure} = 28.76\text{m}^3\text{/day}$$

$$1{:}\ 4.76 = 165.65\text{m}^3 \text{ total influent volume}$$

Influent hydraulic loading would be the greatest factor in the determination of sizing for any treatment ICW system. A high influent hydraulic loading rate of 100m^3/ha/d or even higher would allow for a smaller required land area, but it could yield lower nutrient removal rates. Lower hydraulic loading rates, such as 37m^3/ha/day would increase the size of an ICW, but allow for high retention times and increased rates of nutrient removal. Additionally, dependent upon sizing of the embankments surrounding the wetlands cells, a lower loading rate allows for greater freeboard (fluctuation of hydraulic load) effect. The following sizings are based upon hydraulic loadings with an influent concentration of 600mg/l.

@ 100m^3/ha/day = 1.66 hectares

@ 74m^3/ha/day = 2.238 hectares

@ 37m^3/ha/day = 4.476 hectares

Additionally, if the influent concentration was lowered to 400 mgNH_4/l, then the respective sizings change rather substantially, with a 39% increase in required land area for hydraulic loads of 100, 74 and 37m^3/ha/day.

400mg/l = 1:7 dilution = 230.08m^3/ha/day

@ 100m^3/ha/day = 2.3 hectares

@ 74m^3/ha/day = 3.1 hectares

@ 37m^3/ha/day = 6.2 hectares

With all of these sizings, the factor of a recycling mechanism is not taken into consideration. Dependent upon the method and, for instance, the size and operation of a pump, the dilution rates could be reduced further, perhaps up to 60% or even 75% if the shallow level of water in the latter half of the cells was maintained. Recycling has a noticeable effect on the running of the system as a whole by have several advantages:

- Increase hydraulic retention times
- Decrease the volume of water needed for dilution of influent material
- Addition of partially-nitrified water to the first cell
- Reduced outflow volumes
- Decrease required land-area needed for ICW

The reduction in land-area required for the ICW as a result of a recycling mechanism would be highly advantageous to farmers/landowners who have limited land available to them, or indeed the land that they have is already within or close to its nitrates limit of 170kg/ha/yr. If the stocking rate to land of organic nitrogen is already high (e.g. 120kg/ha/yr), then the land has a capacity for a further 50kg/ha/yr organic-N. The value of 4.2kg N/m^3 (SI No. 160) gives a value of $11.9m^3$ influent/ha. If a piggery is to keep within the limits of the Nitrates Directive, a 500 sow integrated unit would require 40.48 hectares at the minimum dedicated to landspreading. This may indeed include cropland and grazing land, but many piggeries in Ireland would not have this land available to them directly and would incur considerable costs over the duration of a year due to transport to spreadlands (Nolan *et al* 2012).

4.9 Shortcomings of the project

The meso-scale systems proved to be very robust in their application throughout severe weather conditions repeatedly. They were easy to work with, to source and to construct. There were several smaller issues that were discovered throughout the sampling process that, should another be constructed or indeed the same system be used again in the future, would need to be addressed in order to maximise their performance and remove any difficulties or shortcoming with the system as a whole.

In hindsight, the influent concentration levels should be applied at a significantly higher rate than what was initially decided upon in this system. Whilst the value chosen was taken from the literature available at the time (Scholz *et al*., 2007), 100 and 200 mg/l ammonia-N are erring on the side of caution that does not challenge the system. After sampling, recording data, observing and maintaining the site as a whole throughout the 18-month period of fieldwork, the results, and indeed reactions of the systems, suggest that the experimental design could tolerate an influent concentration level of 400 mg/l ammonia-N and perhaps even higher.

The operation of the meso-scale ICW system for a substantially longer period of time would have given a far greater range of data that the single annual cycle that was recorded. Whilst this initial 12 months had sampling limitations due to funding restrictions and project deadlines with the Stimulus Project, the systems themselves could have been kept operational throughout the remaining time available. Having this unique system operational for a longer period of time would have given the chance to compare any potential seasonal variation, and also to examine what the effects seen during the harsh weather in 2009/2010 were in comparison to a milder winter (2010/2012) and to see if these were replicated in the following year. However, this extended sampling was simply not applicable or viable due to funding restrictions. Sampling had only been performed on cells 1 and 2 and on the sampling tanks due to the cost of analysis for this volume of samples: roughly 32 samples a week for 18 months. Inclusion of cells 2 and 3 would have doubled this number and whilst basic nutrient analysis may have been feasible, BOD_5 analysis would have been completely impractical as the water chemistry laboratory in Johnstown Castle, Wexford would have been unable to deal with that volume of samples and the cost of doing them would have meant the dropping of other parameters.

A specific ammonia toxicity test in *Glyceria maxima* was to be carried out in 2011, but due to time and funding constraints, this was never achieved. The meso-scale systems did suffer a sustained accidental discharge from the storage tanks over a period of days where the influent ammonia-N concentrations exceeded 9,000 mg/l. The results from this event were not systematically recorded as it occurred outside of the final 12-month sampling period. However, the event provided the opportunity to take samples for analysis in order to garner supplementary information on the health of the plants and systems. The HNL systems received up to 16,000 mg NH_3/l and only the first cell of each system displayed any negative effects as a result, while cell 2, 3 and 4 showed enhanced growth and response from the additional nutrients available. Higher influent concentrations would have stressed the systems further than they were and could have potentially had a substantial difference on the recorded results. Pushing the meso-scale systems beyond 'common' operational concentrations, typically 100 mg/l NH_3, would have allowed for examination of much higher concentrations and which may have resulted in significant differences between the various treatments. Having influent concentrations of 300mg/l and 500mg/l would have pushed the systems more than their respective 100mg/l and 200mg/l of NH3. It is important to keep in mind that this is speculative and the before a system could be trialled with such concentrations, a specific ammonia toxicity test would have to be carried out.

Alongside operational parameters, other lesser shortcomings were the equipment that were used as part of the siphon traps and the 5th cells that were introduced into the systems later in the fieldwork. The siphon traps were added after the initial test-run of the systems after construction and these should be included in the initial design. Throughout the project, the siphon traps addressed any siphoning problems that had been encountered upon initial start-up. There had been brief issues with blockages with the gravity-fed lines from the siphon traps in certain systems, but this only ever occurred after periods of extremely cold weather where the low temperatures had caused particulate matter to clump in the smaller 6mm hose and block. When such blockages were recorded, these were cleaned and tested immediately.

Several shortcomings are also noteworthy in regard to the sampling methodology and indeed range of sampled parameters. Despite the financial aid given by the Teagasc Stimulus Funding scheme, the original grant was cut by 66% after being processed. This placed significant restrictions on both what was sampled and how frequently they have been sampled. The suite of nutrient parameters that were measured is identical to that which is performed on full-scale ICW systems in Co. Waterford in Ireland. However, these could have been expanded to include a range of additional test parameters, such as atmospheric nitrogen release, ammonia volatilisation and greenhouse gas emissions.

The overflow containers that were added to the system were not utilized correctly throughout the sampling period and with site visits and recording being done only once per week, vital data was lost when heavy rainfall event occurred resulting in large volumes of outflow. Simple flow metering hardware at the end of each system would have provided accurate data throughout the 18-month period and this could have been assigned to the effluent concentration levels. Data such as this could have given accurate information highlighting that while there is a discharge from scaled-up ICW systems during heavy rainfall events, the effluent is diluted by the rainfall and even large discharges would not pose a threat to receiving water bodies.

The examination of greenhouse gas (GHG) emissions from the meso-scale systems was another topic that had been both considered and indeed planned and organised, but something that failed to materialise. The investigation into the release of GHG had been broached by a Masters student from University College Cork (UCC), and some initial design and preparation work had been completed. Test frames had been constructed for the meso-scale cells and had been fitted, but after initial tests their design needed adjustment and after several weeks, contact was lost with the student and the research was considered halted. No further contact had been made with the Masters student, despite efforts to contact them.

4.10 Applications

The use of meso-scale systems such as these, which lean heavily on the work of Herb Bormann (1987), allow for their use in a range of research applications where a pilot scheme is not a viable or practical option for the designer/researcher. Additionally, the results from this study could have significant impacts upon the design, construction and operation of full-scale ICW or other constructed wetland approaches that are dealing with piggery wastewaters. The following applications are discussed.

4.10.1 Research application

Across the vast majority of fields of research, replication and reliability of results is paramount in creating a dataset that the researcher/operator can rely upon to be accurate and that any potential recorded anomalies are easily accounted for. Being able to create a highly replicated system on a scale that surpasses that of a lab-scale system, in terms of real world imitation whilst being able to construct at a relatively low cost, can be immensely practical and cost-saving. This meso-scale approach has the characteristics of an open, biological system that is fully interacting with its surroundings, which can give the researcher/operator greater reliability in the 'up-scaling' of any design features to be implemented in a larger (pilot- or full-scale) system. Scaling up to a large system from a lab-scale where the environment is very controlled and does not take into account the potential interactions with the surrounding environment, can potentially lead to unexpected changes in terms of performance and stability. There are complications with maintaining this 'open-world' approach with the meso-scale because of the vast array of interactions that are occurring with the system. Additional influent from rainfall, detritus from flora and fauna surrounding the site and indeed from passing wildlife as well as ever-changing climatic conditions, all have an effect on the performance, stability and longevity of a small-scale system as this. However, with the ability to create such a highly replicated series of systems and tests on a much smaller budget than many larger pilot-scale systems, as well as potentially expensive lab-scale systems that may require expensive equipment, could be a very welcome option to many researchers. Most pilot-scale systems would be based upon literature where similar test have been performed, or indeed full-scale systems (regardless of field of research, be it wetland or otherwise), or are based upon successful lab-scale experimentation. The ICW design has not been examined in a lab-scale before, only on a pilot and test scale when the concept was being developed in Co. Waterford, Ireland (Harrington and Ryder, 2002). Pilot-scale constructed wetlands have been used for research purposes around the world (heavily in the USA especially) and these systems are often repeatedly used with differing variables and operational procedures. A review of the literature conducted during the conceptualisation, design, construction and operation of the meso-scale ICW in Moorepark, did not identify other meso-scale systems of a similar nature.

5. Conclusion

5.1 Key operations of the meso-scale ICW

The findings of this research indicate for the first time that effluent recycling within ICW treating piggery wastewater is likely to lead to better outflow concentrations of most commonly observed pollutants such as ammonia nitrogen. While the use of a recycling mechanism is seen in other treatment and constructed wetlands around the world (Kantawanichkul, 2002), it has previously not been commonly used in the ICW approach. High flow rate treatment led to the meso-scale wetland systems being partially overloaded, particularly by nitrogen species. This is of specific relevance to piggeries that would have a restricted land area available for the construction of an ICW for wastewater treatment and also have a high wastewater output from their facilities. Balancing the input into an ICW system is crucial to its successful operation. Nitrification performed better in summer compared to winter, which made direct performance comparisons between the low and high flow rate phases difficult. However, considering that all systems with different test conditions were operated in parallel, it is justified to compare system performances relative to each other. In general, low flow rates in comparison to high flow rates led to better system performance. A cost–benefit analysis based on ammonia-nitrogen loads retained indicated for the first time that all ICW systems performed similar regardless of treatment mode (i.e. nutrient and hydraulic loads and effluent recycling). However, doubling of the flow rate is likely to be most economic, especially when considering the quantities of wastewater that can be treated by a wetland even if outflow concentrations for ammonia-nitrogen are slightly higher than for operational modes with a standard flow rate. For small piggeries with sufficient land, the proposed ICW system is a viable, sustainable and cost-effective alternative to anaerobic digestion, bio-membranes and other standard wastewater treatment methods such as the activated sludge process (Appendix 2, Bioresource Technology journal paper).

5.2 The use of ICW as a viable option for piggeries

The use of the ICW concept in the treatment of separated swine wastewaters has been shown to be highly effective in reducing nutrient concentrations. In light of the Nitrates and Water Framework Directives, considering their low-cost and low-maintenance design, the ICW approach could prove to be highly advantageous to piggeries in Ireland, especially where available spreadlands are limited. The nutrient removal rates seen in this meso-scale experiment show that there is a strong potential for the experimental design to be scale-up for application within the ICW concept to be applied to commercial units. A larger, pilot scheme could be used to examine the possibility of combining the different operational parameters into a single system to improve performance and efficiency.

Additional testing of hydraulic residence time and specific ammonia toxicity for specific native macrophytes should also be examined (section 4.8). Despite the relatively high land-take that ICW systems have in comparison to conventional wastewater treatment approaches, their increased nutrient removal rates, additional benefits and 'total' treatment approach to wastewater, coupled with lower construction and operational costs could make them a viable alternative approach for piggeries in Ireland (Appendix 1, Hydrobiologia journal paper).

5.3 The use of the meso-scale design as a research tool

The use of this meso-scale 'sandbox' approach for examination of ICW treatments allows the user a great degree of control over what is essentially an open biological system. The wetlands interacted with their surrounding environment, but all the key design parameters were easily controlled.

External factors such as meteorological conditions can easily be recorded for the immediate area and their effect on the systems can be deduced from the systems themselves as well as the recorded results. The land area required in comparison to bench-scale and indeed some pilot-schemes is significant; however, when viewed in light of the amount of replication that it allows for, the cost is easily acceptable. The ability to compare a larger number of replicates ensures reliability in the recorded data from each system as well as each treatment operation. Recorded data points that were significantly different to other replicate samples were easily identified and dealt with. This also has a significant impact on the reliability of any statistical analysis performed on the recorded data set, for which replication is essential (Appendix 3, Journal of Environmental Science and Health journal paper).

The use of natural, restored or constructed wetlands in the treatment of wastewaters is not a new practice (Knight *et al*., 2000). Their inherent ability to hold and treat wastewater of varying types, strengths and constituents has been adopted for the treatment of various activities including municipal, industrial and agricultural (Kadlec, 1995). Due to their broad spectrum of applications, the mechanics, operations and designs of wetlands have been examined in depth. The USA based their original guidance documents on empirical formulae and set parameters attempting to define a biological system (USEPA, 2000). These original documents laid the groundwork for more research that was to come, utilising their original designs and adapting them to suit more specific or broader uses.

Investigations into the use of constructed wetlands have shown their suitability in a wide range of climates, and demonstrate their effective treatment of wastewaters regardless of climatic conditions. Though government-supported documents are few, they are becoming more available and within Europe there have been several iterations of designs and guidelines (Carty *et al*., 2008; DoEHLG, 2010). With the establishment of the EU and the implementation of the Nitrates Directive, the use of constructed wetlands achieved more attention from researchers and the potential of wetlands to deal with the more troublesome of wastewaters, such as piggery wastewater, was explored. Piggery wastewater poses a distinct problem with its exceptionally high Nitrogen concentrations and varying composition due to the nature of the feedstocks used in the industry. Limiting factors for the effective treatment/disposal of piggery wastewaters, are that of available land for land-spreading, high effluent volumes and limits imposed by the Nitrates Directive on the application of nitrogen onto spreadlands. The combination of constructed wetlands and pre-treatment (settling tanks, dilution and solids removal) can enable treatment wetlands to overcome the obstacle of high nutrient levels and influent variability. Combined with newer designs and operations, this could allow for constructed wetlands to become a viable alternative method to piggery wastewater treatment.

The meso-scale approach upon which this research was founded had not been used before in conjunction with the ICW approach to design and operation. Whilst replicated pilot-schemes and designs have been used in the past, few have implemented such a high level of replication or range of operations examined in parallel with one another. This high degree of replication, ease of operational use, easy observation of fluctuations and practical construction and maintenance is unique, certainly in Ireland and possibly globally. Ensuring that the meso-scale systems were used outdoors and run in such a way that they are interacting constantly with the environment around them as open, biological systems allows for a greater degree of reliability in any hypothesis for full-scale implementation based on the recorded results from the study. They fill a niche between bench/lab-scale and pilot-scale which, based upon the results and experiences gathered from this research project, is a practical and highly adaptable test-bed for future use.

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Appendix 1: Bioresource Technology Paper

Appendix 2: Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental Engineering Paper

Appendix 3: Hydrobiologia Paper

Appendix 4: Quality Controlled data from fieldwork.